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Review article

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Nitrogen management during decarbonization

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Abstract

Decarbonization is crucial to combat climate change. However, some decarbonization strategies could profoundly impact the nitrogen cycle. In this Review, we explore the nitrogen requirements of five major decarbonization strategies to reveal the complex interconnections between the carbon and nitrogen cycles and identify opportunities to enhance their mutually sustainable management. Some decarbonization strategies require substantial new nitrogen production, potentially leading to increased nutrient pollution and exacerbation of eutrophication in aquatic systems. For example, the strategy of substituting 44% of fossil fuels used in marine shipping with ammonia-based fuels could reduce CO_2 emissions by up to 0.38 Gt CO_2-eq yr^{-1} but would require a corresponding increase in new nitrogen synthesis of 212 Tg N yr^{-1}. Similarly, using biofuels to achieve 0.7 ± 0.3 Gt CO_2-eq yr^{-1} mitigation would require new nitrogen inputs to croplands of 21–42 Tg N yr^{-1}. To avoid increasing nitrogen losses and exacerbating eutrophication, decarbonization efforts should be designed to provide carbon–nitrogen co-benefits. Reducing the use of carbon-intensive synthetic nitrogen fertilizer is one example that can simultaneously reduce both nitrogen inputs by 14 Tg N yr^{-1} and CO_2 emissions by 0.04 (0.03–0.06) Gt CO_2-eq yr^{-1}. Future research should guide decarbonization efforts to mitigate eutrophication and enhance nitrogen use efficiency in agriculture, food and energy systems.

Sections

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Introduction

Climate change and human alteration of global nutrient cycles are two environmental threats that have already exceeded their planetary boundaries and require urgent action to bring them back within safe operating space[1–3] (Fig. 1). Climate change, largely driven by anthropogenic greenhouse gas (GHG) emissions, has led to global warming and a range of environmental issues, from more frequent and intense weather events to sea level rise[4]. Anthropogenic GHG emissions, dominated by carbon dioxide (CO_2), reached 59 ± 6.6 Gt CO_2 equivalent (CO_2-eq) in 2019 (ref. 5) (Fig. 1a). Unless emissions are reduced, the mean global temperature could exceed 1.5 °C global warming relative to pre-industrial temperatures, which could trigger multiple climate tipping points[6]. Thus, there is an urgent need to adopt decarbonization strategies to mitigate climate change.

However, some decarbonization strategies, such as swapping fossil fuels with ammonia-based fuels or biofuels, could increase nitrogen (N) inputs into the environment. Such strategies could enhance N-polluting human activities and contribute to other environmental issues, including eutrophication of waterways[7], declining regional air quality, global ozone depletion and climate change[8]. Thus, strategies should simultaneously abate GHG emissions and nutrient pollution to mitigate both climate change and eutrophication.

Anthropogenic inputs of N to the environment are already nearly double the natural terrestrial and oceanic inputs from N fixation (estimated to be 110 and 140 Tg N yr^{-1}, respectively[9], compared to 256 Tg N yr^{-1} of human inputs[10]; Fig. 1b). Excessive N inputs to waterways are the primary driver of eutrophication[7], as elevated N concentrations trigger proliferation of algal biomass that can lead to dissolved oxygen depletion and toxin production[11]. Eutrophication impacts threaten water security and ecosystem and human health. For example, the annual damage cost of eutrophication in the USA is estimated over US$4 billion (ref. 12). Eutrophication is already globally prevalent, intensifying in most lakes since the 1980s (ref. 12) and impacting coastal and open oceans[13,14]. Therefore, unsustainable management of N use in decarbonization strategies could have devastating impacts[10].

The increasing feasibility of producing ammonia (NH_3) using renewable energy has led to growing enthusiasm for using NH_3 as a fuel to decarbonize the shipping sector[15] (see The Cool Down). But replacing fossil fuels in shipping with NH_3 would more than triple NH_3 demand and production, which is already beyond the planetary boundary for human disturbance of N cycles[1,2,16]. However, some decarbonization strategies have the potential to provide co-benefits for addressing both climate change and eutrophication through simultaneously reducing emissions of CO_2 and N species[17,18]. For example, electrification of energy supplies and transitioning to renewable fuels will probably reduce reliance on fossil fuel combustion and, thus, reduce associated emissions of nitrogen oxide (NO_x) and NH_3 (ref. 19). Understanding the impacts of decarbonization efforts on the global N cycle prior to implementation will help avoid exchanging one pollutant and set of environmental impacts for another and to promote synergies of multi-pollutant reductions.

In this Review, we summarize the environmental impacts of excessive human N use, with a focus on eutrophication, and explore the potential synergistic and antagonistic impacts on carbon (C) and N emissions from five major decarbonization strategies. These strategies include reducing C-intensive synthetic N fertilizer production, growing plant biomass as a renewable energy source, using NH_3 as C-free fuel, sequestering C in agricultural soil, and intensifying crop yield to reduce pressure for land use change and deforestation. Through greater consideration and synergistic management of the interconnected C and N cycles, decarbonization efforts can simultaneously address climate change and avoid exacerbating eutrophication.

Nitrogen impacts on the environment

N is critical for producing the biological molecules necessary for life, such as proteins and nucleic acids[20]. Although N is the most abundant element in the atmosphere, it mostly exists as inert dinitrogen (N_2) which cannot be directly utilized by most living organisms without first being converted into reactive forms of nitrogen (N_r), such as NH_3 (Fig. 2). In nature, biological N fixation by organisms, such as N-fixing bacteria that are either free living or in symbiosis with leguminous plants, facilitates the conversion of N_2 to N_r. Globally, N-fixation has been adding N_r at the rate of about 250 Tg N yr^{-1}, yet N_r availability remains a key limiting factor for global primary production[9]. The invention of the Haber–Bosch process in the early twentieth century enabled N fixation through an industrial process and was used to produce synthetic N fertilizer to alleviate N limitation and enhance crop yields[21]. Ever since, N_r input to the Earth system by human activities has increased exponentially, fast approaching the N_r input by biological N fixation processes in nature and disrupting the global N cycle[20,22] (Fig. 1b).

The drastic increase of synthetic N fertilizer use boosted agricultural productivity, but it has also led to a wide range of environmental issues. Only about half of the N applied in crop production is assimilated into harvested crops[23], with the remaining – termed N surplus[24] – accumulating in the environment. N surplus can enter groundwater and surface waters through leaching and runoff, causing eutrophication and contamination of drinking water[25]. Eutrophication degrades the environment as it depletes oxygen in the water creating hypoxic zones lethal to aquatic life[11]. Algae that form eutrophic blooms can also generate toxins that can cause extensive biodiversity loss in aquatic ecosystems[11]. N surplus can also be emitted as NH_3 and NO_x to the atmosphere where it contributes to regional air pollution through the formation of fine particulates, nitrogen dioxide and ozone, and poses risks to human respiratory health[26]. Upon deposition, atmospheric N emissions can disrupt the nutrient balance in natural ecosystems and potentially cause declines in plant diversity, habitat degradation[27] and eutrophication[28]. N surplus can also be converted back to N_2 through denitrification processes in both terrestrial and aquatic ecosystems, with a fraction being emitted as N_2O, which is currently the most abundant ozone-depleting substance in the stratosphere and a prominent greenhouse gas[29]. Some N surpluses could be retained in soil leading to a buildup in soil N stocks that could potentially enhance future crop yields, but this retention also renders these N pools vulnerable to being emitted in various forms over subsequent years[16].

Over 70% of the harvested N in crop products is used as feed for livestock production, which is even less efficient in converting N to food products than most plant-based human food[30]. This inefficiency results in a high proportion of residual N in manure[31], which is readily lost to the environment through leaching, runoff, and NH_3 and N_2O emissions[11]. Recycling manure back to cropland and pasture could potentially offset up to 70% of synthetic N fertilizer demand[31] and improve the efficiency of N use in agricultural production systems. However, broad implementation of this practice has been limited by economic and logistical challenges, such as the cost of transporting manure that has high water content[32–35].

The production process and transportation of synthetic N fertilizer impacts the environment and contributes to climate change. Synthetic

N fertilizer production is an energy-intensive process, currently accounting for 2% of global energy consumption[36], about 4% of global natural gas use, and 1% of GHG emissions[37,38]. In 2019, synthetic fertilizer production was responsible for about 0.45 Gt CO_2-eq yr^{-1} of emissions, primarily from conventional industrial NH_3 production using the fossil fuel-based and energy-intensive Haber–Bosch process[36]. Transportation of fertilizers adds another 0.03 Gt CO_2-eq yr^{-1} in emissions[38]. The CO_2-eq of combined GHG emissions from the production and transport of synthetic N fertilizer, which are dominated by CO_2, is similar to that of the emissions resulting from the application of synthetic N to cropland soils, which are dominated by N_2O emissions estimated to be 0.63–0.66 Gt CO_2-eq yr^{-1} (refs. 37,38).

Overall, human activities that convert N_2 to N_r, predominantly through the Haber–Bosch process, have been pivotal in enhancing agricultural productivity and food security. Yet, industrial production of N_r has driven the exponential growth of N input to the environment that has led to an increasingly eutrophic world[20,22]. Enhanced N inputs have posed risks to ecosystem and human health, owing to not only the eutrophication of water bodies but also the release of air pollutants and the exacerbation of climate change and stratospheric ozone depletion. Moreover, N_r introduced by human activities can lead to multiple environmental impacts as it changes forms, such as microbial processes within the natural N cycle that oxidize ammonium to nitrate and reduce nitrate to nitrous oxide, each with its own environmental impacts, a phenomenon known as the N cascade effect[20]. Therefore, it is critical to manage and track N_r introduced by human activities, its loss to the environment, its transformations between different N_r species, and its fate.

Decarbonization strategies and nitrogen

The C and N cycles are intimately linked (Fig. 2). There are five major decarbonization strategies (DSs) that impact human-driven N inputs and losses in the Earth system (Table 1, Fig. 3; Supplementary Table 1). This section synthesizes quantitative estimates of the C mitigation potential and the direct effects of decarbonization strategies on CO_2 mitigation, and the corresponding changes in N inputs and N losses for the five major DSs. Here, N input refers to N_r that is converted from inert N_2 by human activities, which is synonymous with new N_r introduced into the environment by humans. N inputs are dominated by synthetic fertilizer N production through the Haber–Bosch process, but they also include N_r fixed by crops and N_r emitted from fossil fuel burning (Fig. 1b). Effects on N_2O emissions and perhaps CH_4 emissions would also probably occur, but the estimation of net GHG emissions is excluded here owing to the large uncertainties of N_2O and CH_4 emission factors and to the complexities of C–N interactions.

Reducing carbon-intensive fertilizer production

Reducing CO_2 emissions from synthetic N fertilizer production (DS1) involves two types of efforts with very different impacts on N inputs and losses[39] (Supplementary Fig. 1). One effort is to reduce the C intensity of NH_3 production (Fig. 3a; DS1a). Since 2009, the emission intensity of N fertilizer production has decreased by 12% (ref. 36), but further reductions in emission intensity from conventional production using the Haber–Bosch method are expected to have a relatively modest impact on energy and emissions savings[40]. More transformative measures are required, such as integrating C capture and storage with continued fossil fuel use (called blue NH_3), using carbon-free electricity for water electrolysis (called green NH_3), and using biomass as a feedstock for the Haber–Bosch process[39,41]. These methods could reduce NH_3 production emissions by 95%, and net-zero C emissions could be achieved by offsetting residual emissions with CO_2 removal technologies[39]. However, these technologies are limited by scale and their high investment and operation costs, with environmental trade-offs related to energy, land, water, and biomass usage[42] that exacerbate land and water scarcity[41,43]. Nevertheless, major efforts to produce NH_3 with renewable energy sources or with C capture and storage are being implemented[44–48]. In addition, new catalysts for synthesizing NH_3 at ambient temperature and pressure, and photochemical reaction-based catalysts, hold the

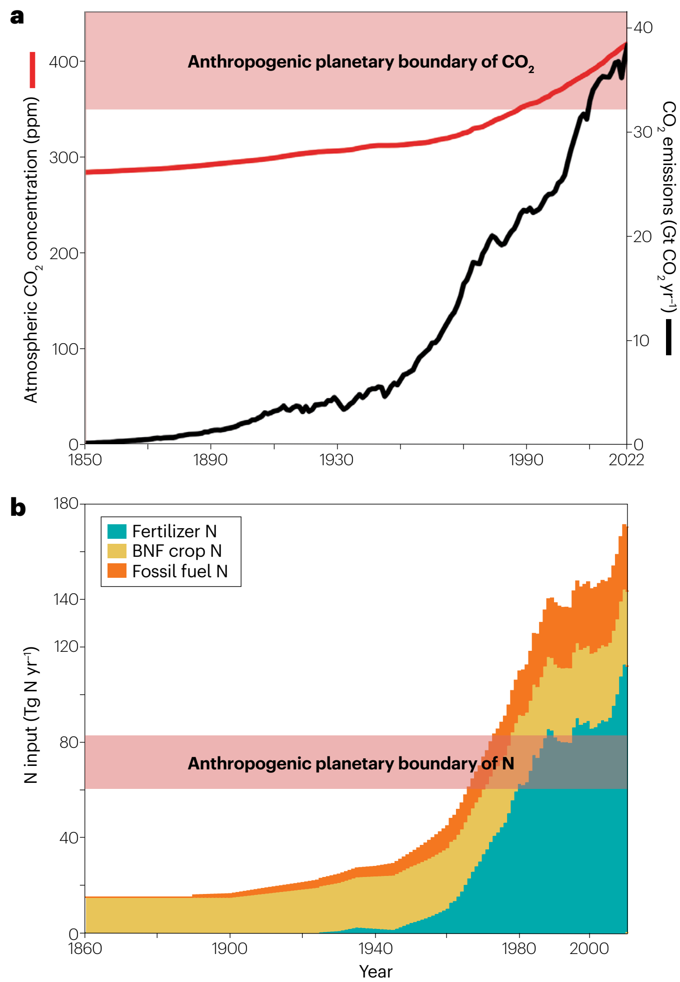

Fig. 1 | Historical trends of CO_2 emissions and nitrogen (N) inputs from human activities. a, Atmospheric CO_2 concentration (solid red curve, ppm) and CO_2 emissions (solid black curve, Gt CO_2) between 1850 and 2022, with atmospheric CO_2 concentration representing the planetary boundary indicator for climate change. **b**, Annual N inputs (Tg N yr^{-1}) from fertilizers (fertilizer N, green), crop biological nitrogen fixation (BNF crop N, yellow) and fossil fuel use derived nitrogen inputs (fossil fuel N, orange) between 1860 and 2015. The planetary boundaries for N input and atmospheric CO_2 concentration are noted as red zones in both panels and are taken from ref. 1; red shaded areas represent the uncertainty range of the planetary boundary. Data for N inputs are taken from ref. 24. CO_2 concentrations are taken from ref. 172 (see National Oceanic and Atmospheric Administration) and CO_2 emissions are taken from refs. 102,173–175. CO_2 emissions and N inputs from human activities have exceeded their planetary boundaries and require urgent mitigation efforts.

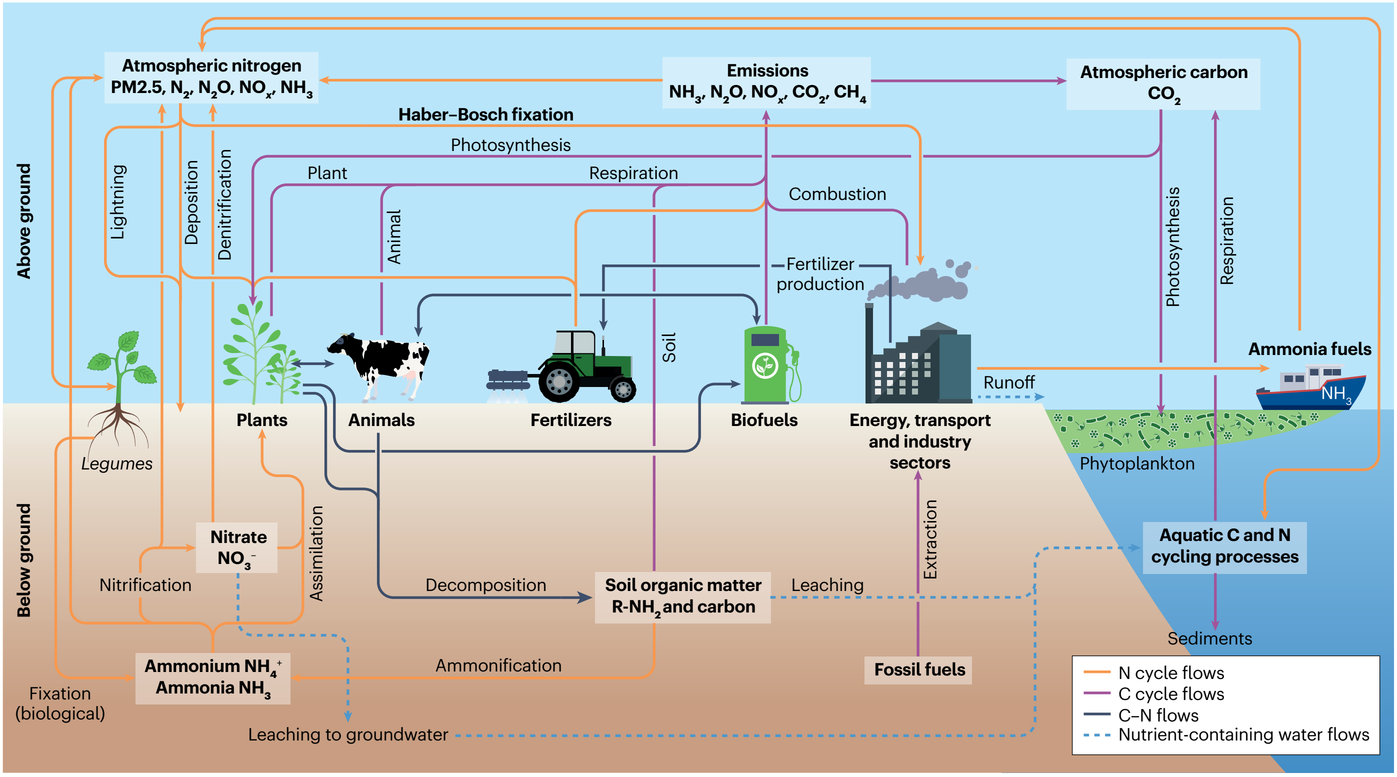

Fig. 2 | Interconnections and human perturbations of the carbon and nitrogen cycle. Purple lines represent carbon cycle flows, orange lines represent nitrogen cycle flows, black lines show carbon–nitrogen flows, and blue dashed lines represent nutrient-containing water flows. The carbon cycle comprises 'photosynthesis', wherein plants convert carbon dioxide (CO_2) and water into glucose and oxygen using sunlight; 'respiration', wherein organisms release CO_2 back into the atmosphere by breaking down organic-C substrates for energy; 'decomposition', wherein bacteria break down dead organic matter to use for their respiration, releasing CO_2 back into the environment; 'combustion', wherein burning of fossil fuels or biomass releases CO_2 into the atmosphere; and 'sedimentation', wherein carbon can be stored in sediments and potentially become fossil fuels over long periods. The nitrogen cycle constitutes 'nitrogen fixation', wherein inert atmospheric dinitrogen (N_2) is reduced to ammonia (NH_3), either biologically such as a bacterial symbiosis with leguminous plant roots or through industrial processes (for example, the Haber–Bosch process); 'nitrification', wherein NH_3 and ammonium (NH_4^+) are oxidized to nitrite (NO_2^-) and then to nitrate (NO_3^-); 'assimilation', wherein plants, animals and microbes take up NO_3^- and NH_4^+ for anabolism, which then gets converted to soil organic matter through 'decomposition' upon their death; 'ammonification', wherein organic nitrogen gets converted into NH_4^+ by decomposers; and 'denitrification', wherein NO_3^- is reduced to nitric oxide, nitrous oxide and dinitrogen gas emitted to the atmosphere. NO_3^- can also 'leach' from terrestrial ecosystems to aquatic ecosystems[10,54,122,123,125,126]. Please note that it is not our intention to illustrate all N and C compounds, and their fates, in this figure, but to instead focus on the most relevant ones to this review.

potential to bypass the energy-intensive Haber–Bosch process[49–51]. Overall, the production capacity of low-C NH_3 is projected to increase from the current 0.02 Mt yr^{-1} to 15 Mt yr^{-1} by 2030 (ref. 42).

Technologies to reduce the C intensity of NH_3 production can support decentralized production, which in turn decreases CO_2 emissions from transportation and improves N resource distribution[52]. These technologies include electric Haber–Bosch, non-thermal plasma-activated N fixation, photocatalytic N reduction and direct electrocatalytic N reduction[52]. Such technologies allow for smaller production facilities, which are more than two orders of magnitude less than the capacities of 2,000–3,000 t NH_3 per day in current large and centralized plants[52]. Although GHG emissions from transporting NH_3 are relatively small (<10% of the emissions from fertilizer production) and so the emission mitigation benefits are modest, localization of fertilizer production offers the added benefits of reducing the transportation costs for farmers, especially in the Global South where distances from production plants are often large[52].

Technologies to reduce the C intensity of NH_3 production hold great potential for mitigating C emissions during fertilizer production and transportation, but their impact on the amount of N used in agricultural production and the associated N loss to the environment remains unclear. For example, distributed systems that provide more reliable year-round fertilizer access could enable better synchronization of N fertilizer application with plant demand and discourage application at times when fertilizer use is inefficient (for example, fall application, which is common in the US Midwest)[53]. The increased availability and accessibility of N fertilizer will also probably increase N inputs in regions with poor infrastructure (for example, sub-Saharan Africa) and possibly promote more fertilizer-intensive crop production or practices[52]. The latter development could be appropriate in regions where increases in domestic food production are needed or where efforts are needed to reduce GHG emissions from land conversion for agricultural expansion[54] (and other co-benefits such as minimizing biodiversity loss)[55]. In addition, distributed green NH_3

production could reduce the reliance of a region on fertilizer imports and make agriculture–food systems more resilient to supply chain shocks[39].

Decreased demand for synthetic N fertilizer would also reduce CO_2 emissions from synthetic N production, which in turn would reduce the overall GHG emissions associated with transportation and application (Fig. 3a, DS1b). This decarbonization strategy aligns with the need to address eutrophication challenges and can be achieved by improving N management on crop and livestock farms, and throughout the supply chain of agriculture–food systems. Estimates suggest that food demand could still be met if the N input to crop production was reduced by 14 Tg yr^{-1} (ref. 16), which would be equivalent to a reduction of 0.04 (0.03–0.06) Gt CO_2 yr^{-1} emissions from fertilizer production, assuming the current C intensity of fertilizer production at 2.52 (2.18–3.91) kg CO_2 per kg N fertilizer[39] (Table 1; Supplementary Table 2).

Improving N use efficiency (NUE) of crop farms through better management of N fertilizer application would provide sustained levels of crop production despite lower N inputs[42]. Many technologies and management practices, including enhanced efficiency fertilizers, cover copping and no-till farming, improve NUE at the plot scale[56–60]. However, there is a limit to how much inputs can be reduced, as N is needed in crop production systems to boost yield and avoid soil N depletion. Another measure to lower N application rates and losses is increasing the use of alternative N sources, such as biological N fixation, manure, anaerobic digester effluent and composts to reduce dependency on synthetic fertilizer N inputs[61]. Soil N-fixing bacteria can convert atmospheric N to NH_3 for crop use[62]. These bacteria are typically associated with legumes such as beans and peas, but ongoing genetic engineering efforts aim to introduce biological N fixation to cereal crops[63,64]. Management practices such as crop rotation and intercropping with legumes offer additional means to enhance biological N fixation and reduce reliance on synthetic fertilizers. Using livestock manure as an N source for crops is another alternative to using synthetic N fertilizers[33,35]; however manure can cause higher N_2O emissions meaning that its net effect on GHG mitigation is uncertain and inconsistent[65]. The net effect of manure application is further complicated by factors such as the emissions associated with manure collection, processing, transportation, and application[37,66]. Additionally, the increasing spatial separation of crop and livestock production reduces the efficiency of recycling manure to cropland[31,67,68]. Finally, improving the efficiency of the agriculture–food system and minimizing waste from farm to table can further reduce demand for fertilized crops[39]. It is estimated that roughly one-third of agricultural products are lost or wasted from farm to table[69,70]. By reducing these losses throughout the supply chain, the need for increased farm production could be alleviated, as could the demand for additional N inputs[39,71].

Growing plants for biofuels

Biofuels are projected to have the potential to mitigate approximately 0.7 ± 0.3 Gt CO_2-eq yr^{-1} by 2030 (ref. 17). Using biofuels as an alternative to fossil fuels (DS2) has become an important element in strategies to mitigate C emissions (Fig. 3b). The production of biofuel in the USA, which is mainly as corn-based ethanol and some biodiesel, has increased sharply from less than 3.8 billion l in 2000 to 59.4 ± 4.5 billion l in 2019 (ref. 72), accounting for approximately

Table 1 | The impacts of major decarbonization strategies on nitrogen (N) use

Decarbonization strategies (DS)	Mitigation potential ($GtCO_2$-eq yr^{-1} or 1,000 $TgCO_2$-eq yr^{-1})	Major impacts on N use	Potential change in N use (Tg N yr^{-1})	Potential change in N loss (Tg N yr^{-1})
DS1: Reducing carbon-intensive fertilizer production	0.43[a]	DS1a: If achieved by reducing carbon-intensity for centralized N production, no impact on N use; the impact of green ammonia on N use is unknown	0 or unknown	0 or unknown
	0.04 (0.03–0.06)[b]	DS1b: Reducing N inputs through better N management can be win–win for CO_2 mitigation and N input reduction	−14 (ref. 16)	−48 (ref. 16)
DS2: Growing plants for biofuels	0.7±0.3[c]	N inputs are needed to grow the plant materials for biofuel	21±9 for corn ethanol; 42±18 for soybean biodiesel[d]	6±3 for corn ethanol; 4±2 for soybean biodiesel[e]
DS3: Using NH_3 as a carbon-free fuel	Up to 0.38[f]	More NH_3 needs to be produced for fuel usage	212[g]	Unknown[h]
DS4: Sequestering carbon in agricultural soil	0.6 (0.4–0.9)[i]	Given the typical C-to-N ratio of soil, soil C fixation is accompanied by N fixation, but whether the fixed N is from additional N inputs or saving the N losses varies	0–14[j]	14–0[j]
DS5: Intensifying crop yields to reduce pressure for land use change	3.4 (2.3–6.4)[k]	Theoretically connected but challenging to quantify	23–512[l]	13–282[l]

[a]Value calculated as 95% reduction from total CO_2 emission from fertilizer production, representing an upper limit of the mitigation potential[39]; the total CO_2 emission from fertilizer production in 2020 is from ref. 36. [b]Value calculated based on the mitigation potential in N input and N surplus noted in ref. 16. [c]Value the mitigation potential of biofuels with a carbon cost at US\$100 t^{-1} CO_2 (ref. 17). [d]These estimates will be more uncertain if to consider all GHG mitigation (for example, including N_2O). [e]Considers that nitrogen use efficiencies (NUE) for corn and soybean are at 70% and 80%, respectively (Supplementary Tables 3 and 4). [f]This value for 44% of the CO_2 emission from shipping sector in 2022, calculated as 44%×0.855 Gt CO_2-eq according to ref. 92. The actual level of CO_2 mitigation will also depend on the quantity of NH_3 used to replace shipping fuel and on the carbon intensity of the NH_3 (Supplementary Table 5). [g]Value calculated based on the NH_3 needed for replacing 44% of the shipping fuel in 2022, equivalent to 44%×11 EJ. [h]Depends on the emission factor of the NH_3 used as fuel owing to leakage and incomplete combustion. [i]Mitigation potential of soil carbon management in croplands with a carbon cost at US\$100 t^{-1} CO_2 (ref. 17). [j]Detailed explanation is available in Supplementary Fig. 5. [k]Mitigation potential of reducing deforestation and degradation with a carbon cost at US\$100 t^{-1} CO_2 (ref. 17). [l]Considers global averaged yield and NUE (Supplementary Fig. 7 and Supplementary Table 7).

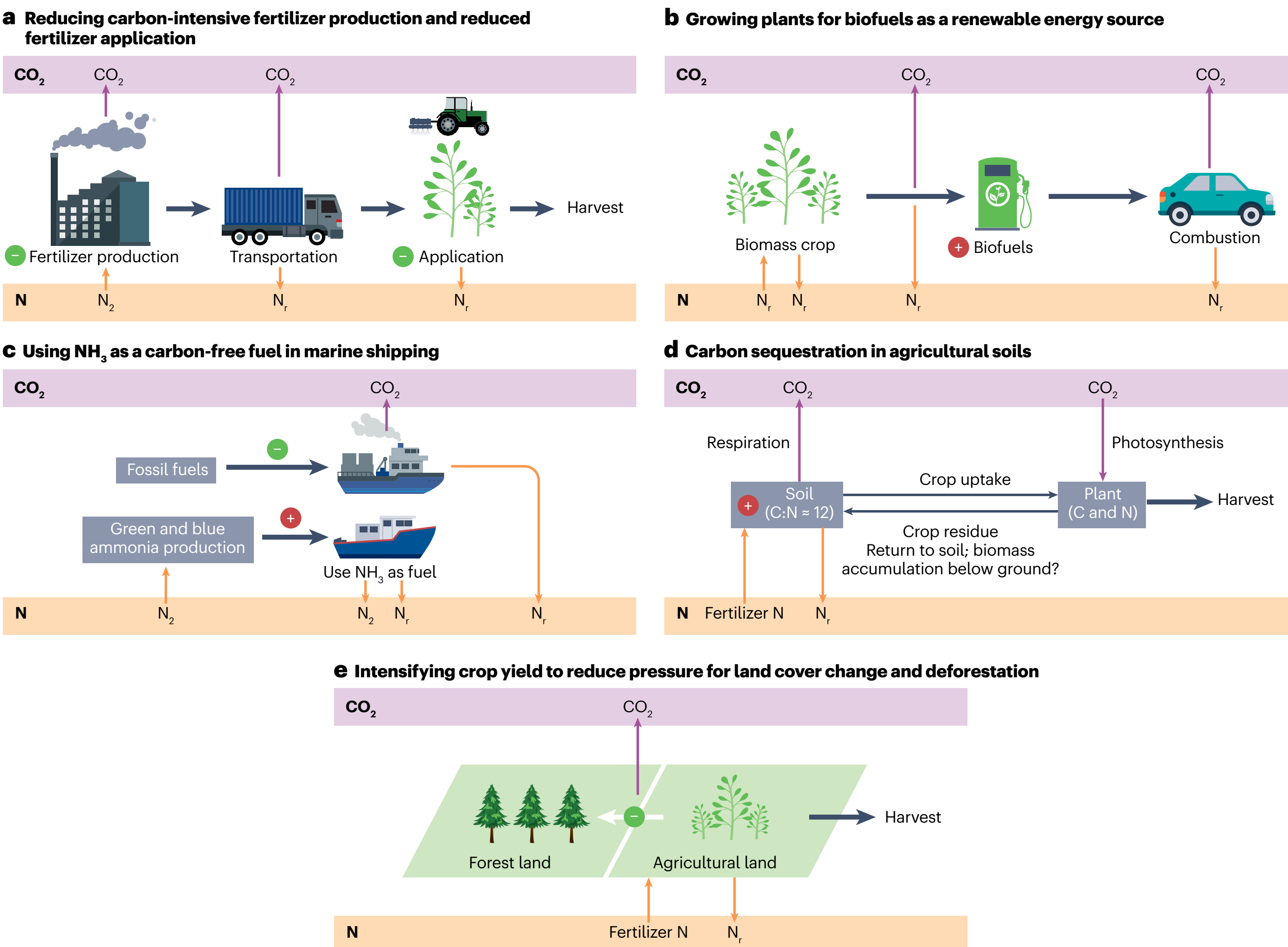

Fig. 3 | Decarbonization strategies (DS) and their impact on nitrogen and carbon cycling. **a**, Reducing carbon-intensive synthetic N fertilizer production and reduced fertilizer application (DS1a and DS1b). **b**, Growing plants for biofuels as a renewable energy source (DS2). **c**, Using NH_3 as a carbon-free fuel in marine shipping (DS3). **d**, Carbon sequestration in agricultural soils (DS4). **e**, Intensifying crop yield to reduce pressure for land use change and deforestation (DS5). Orange arrows indicate major N flows whereas purple arrows indicate CO_2 flows. The minus (green) and plus (red) signs indicate the major intervention points of each strategy. For DS1 (panel **a**), there are two intervention points: the green minus sign on the left indicates DS1a, reducing carbon intensity of fertilizer production, whereas the green minus sign on the right indicates DS1b, reduced fertilizer application. N_2 and N_r represent nitrogen gas and reactive N, respectively. The purple-shaded regions represent impacts on carbon emissions and the orange-shaded regions represent impacts on nitrogen use and emissions. The five major decarbonization opportunities have varying impacts on nitrogen inputs to the Earth system.

half of the global biofuel production[73]. However, substantial uncertainties remain regarding the net impact of biofuels on climate and the environment[73,74].

Three generations of biofuels have been developed using different feedstocks, with the first generation – primarily derived from food crops such as corn, sugarcane and soybean – dominating the market[75]. However, there are concerns over the N demand of these crops and its effects on the N cycle, in addition to concerns regarding land use change, the overall net impact on CO_2 mitigation and potential disruptions to food supply[76–78]. For example, the calculated C intensities of US ethanol production range from 51.4 to 122.1 g CO_2-eq MJ^{-1} (refs. 73,74,79), which is of a similar order as that of gasoline's C intensity of 93.1 g CO_2-eq MJ^{-1} (ref. 79) (Supplementary Table 3). Variation among these reported values predominantly arises from the accounting of GHG emissions related to land use change induced by feedstock crop production, which is a highly uncertain and sensitive driver, and the accounting of downstream recycled co-products, such as soy meal or distillers' grain into livestock production. To provide 1 MJ of biofuel, 0.05 or 0.03 l of first-generation bioethanol or biodiesel is needed respectively (see US Department of Energy), requiring the production of 0.11 kg of corn or 0.03 kg of soybean[80] (Supplementary Table 3). Consequently, if the CO_2 mitigation target of 0.7 ± 0.3 Gt CO_2-eq yr^{-1} is met by corn ethanol or soybean biodiesel alone, then about 21 ± 9 Tg N yr^{-1} or 42 ± 18 Tg N yr^{-1} of N inputs are required, respectively, which leads to 6 ± 3 Tg N yr^{-1} or 4 ± 2 Tg N yr^{-1} increase in N surplus based on an optimistic corn and soybean NUE at 70% and 80%[16] (Table 1; Supplementary Table 4).

Despite the intensive N inputs required to grow biofuel crops, very little N is present in the final biofuel product after distillation[81] and so large amounts of N are available for reuse and recycling. N retained in distiller grains or crushed soy meal has been increasingly used to produce livestock feed and potentially offset the impacts of biofuel production on N inputs to cropland[82,83] (Supplementary Fig. 2). Taking US ethanol production as an example, in 2012, approximately 1.8 Tg N corn was used for biofuel production, requiring about 2.5 Tg N inputs (about 1.9 Tg N input as fertilizer), of which 1.4 Tg N was recovered as feed co-product[16,84] (see US Department of Agriculture Economic Research Service) (Fig. 4), which was used in domestic or international livestock markets[85]. Consequently, it has been suggested that 70%–76% of the N inputs for corn ethanol production should be considered as directly related to ethanol production and that the remainder should be attributed to the impacts of feed co-product, thereby reducing the N intensity of ethanol production[83]. Nevertheless, how N removed from the biofuel production process is utilized and accounted for is critical for determining the overall impacts of biofuel production on the N cycle.

The second generation of biofuels is sourced from non-food crops and agricultural residues, such as corn stover, sugarcane bagasse, wood and grass, and the third generation is sourced from algae. Although removing agricultural residues can disrupt the C and N balance in soils, both second-generation and third-generation biofuels are probably less disruptive to the N cycle than first-generation biofuels derived from food crops[72,86]. Algae are viewed as an ideal third-generation biofuel feedstock[87], as algal carbohydrates can be converted into bioethanol, and algal oils can be used to make biodiesel[88], but its growth requires N inputs. When managed effectively, algal biomass extraction could also serve as a mechanism to remove excess N from polluted aquatic ecosystems and address eutrophication issues[89].

Overall, the contribution of biofuels to fueling transportation fleets in the future, such as larger trucks and airplanes, is projected to increase[90]. Efforts to minimize cropland expansion and optimize nutrient use efficiency in crop cultivation will be key for optimizing biofuel GHG reduction potential and minimizing its potential impacts on eutrophication. However, given the relatively rapid rise in purchases of electric cars in the light and medium duty vehicular fleets across the USA, Europe and China, whether biofuels will be used for those classes of vehicles is unclear, which may limit the widespread uptake and climate mitigation of efficacy of biofuels (see International Energy Agency).

Using NH_3 as a carbon-free fuel

The potential to produce blue or green NH_3 with little to no CO_2 emissions enables NH_3 to be used as a low-GHG transportable energy and hydrogen carrier for a variety of uses throughout the energy and industrial sectors (DS3; Supplementary Fig. 3). By 2030, Japan aims to import low-GHG NH_3 to replace two-thirds of its energy demand for industrial energy, feedstocks and power, which currently rely largely on fossil fuels imported from Australia and Saudi Arabia[91]. Since 2021, there has been increasing focus on the potential of using NH_3 to decarbonize maritime shipping (Fig. 3c). Prototypes of NH_3-powered ships are under construction, for example, in China and Norway[15] (see The Cool Down), and several multi-billion-dollar NH_3 production plants with capacities of more than 1 million t per year are either planned or in development in Saudi Arabia, Oman and USA (see Nutrien). Projections suggest that the contribution of NH_3-based fuels in the maritime shipping industry will grow from 0% in 2024 to 44% in 2050 (equivalent to about 5 EJ yr^{-1}), which would lead to reductions of up to 0.38 Gt CO_2-eq yr^{-1} and contribute to the GHG mitigation goals of the industry[92] (Supplementary Table 5 and Supplementary Fig. 4).

The technical and economic basis for using NH_3 as a maritime fuel is largely grounded in its physical properties. NH_3 is carbon-free and combustible, has no direct global warming potential, and has energy densities (by volume and mass) reasonably practical for maritime transport[93] (Supplementary Table 6). The production costs for NH_3 and its fuel-cycle thermodynamic energy requirements outperform those of fuels produced with electricity from renewable sources, whose C is produced from CO_2 (ref. 94). Using NH_3 as a fuel also has advantages relative to liquefied hydrogen. Liquefied hydrogen has comparatively higher energy requirements, higher costs of liquefaction, greater explosiveness, and high boil-off losses and other fugitive emissions than NH_3 (ref. 95). Fugitive hydrogen emissions also have indirect global warming potential[96].

However, a 44% replacement of maritime fuels with NH_3 would entail an increase of 212 Tg N yr^{-1} in global NH_3 production[10], implying substantial increases in global reactive N production for any large-scale implementation of NH_3 in energy and industry (Table 1). Ideally, such uses of NH_3 would return N to its inert atmospheric form (N_2) prior to being released back into the environment. However, at these scales even modest fugitive emission rates of NH_3, as well as secondary emissions of NO_x and N_2O, could cause deleterious environmental impacts, including air pollution, eutrophication, climate change, and stratospheric ozone depletion[18]. NH_3 leakage from existing production and transportation facilities remains poorly quantified[97], but it has been suggested that the leakage rate for NH_3 could range from 0.3 to 2.5%, drawing parallels with methane[98]. Additionally, 0.2–2.5% of NH_3 might be emitted as NO_x or N_2O during combustion[98]. However, direct measurements and observational data are needed to validate and refine these estimates.

Many-fold increases in NH_3 demand are expected to increase NH_3 prices, which could also affect the price of fertilizer and food, impacting food security and the profitability of agricultural production[52,91].

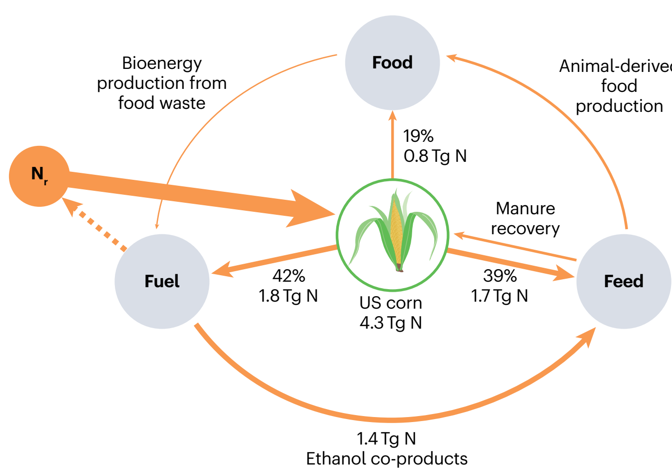

Fig. 4 | The food–feed–fuel triangle. The case for US corn as an example of the competing uses of crop products between feed, fuel and food (including food, seed, exports and other uses). The orange arrows demonstrate nitrogen flows and the numbers represent US corn usage share and amount for feed, fuel and food usage[16,84] (see US Department of Agriculture Economic Research Service). The dashed orange arrow represents the energy requirements of the production and transportation of synthetic nitrogen fertilizer. A very limited amount of crop N products are used for direct food consumption.

In addition to the first-order equilibrium price impacts from a demand shock, the anticipated adoption of green and blue NH_3 as the standard production technologies could put further upward pressure on prices[44]. The 2022 production costs of green NH_3 are two to three times higher than the dominant NH_3 production technology, steam methane reforming[99], though future scale-up and consequent learning-by-doing in electrolysis and renewable electricity generation are expected to substantially bring down these costs[52,99,100].

The markets for NH_3 for fuel and fertilizer could be sufficiently different from one another to insulate agricultural producers from potential price impacts owing to marine shipping fuel demand, but that remains uncertain. Without robust regulation across country borders, the feasibility of separating two different markets for the same physical commodity is probably limited[52]. Higher fertilizer prices, if they occur, would increase costs and reduce profit margins for agricultural producers, and might also encourage more efficient use of fertilizer, leading to reductions in system-wide nutrient runoff and N_2O emissions and the associated impacts on eutrophication. The potential impacts of these simultaneous and notable changes to NH_3 markets will vary greatly by region and over time. The large uncertainties in how NH_3 fuels may develop make it crucial to predict market impacts, identify key causal mechanisms and N cycle impacts, and develop strategies to mitigate adverse outcomes.

Sequestering carbon in agricultural soils

Sequestering atmospheric C in agricultural soils (DS4) has gained considerable attention as a climate change mitigation strategy[101] (see United Nations Department of Economic and Social Affairs). Soils store more C than aboveground vegetation and the atmosphere combined[102], but the legacy of global agriculture is estimated to have resulted in the loss of 133 Pg C from the top 2 m of soils converted to agriculture[103]. Tillage makes soil more vulnerable to erosion and exposes soil organic matter to oxygen and microbial enzymes that increase rates of decomposition[104]. To reverse this historic soil C debt, a variety of soil C markets are developing to promote soil C sequestration through a variety of best management practices, such as reduced tillage and planting winter cover crops[105]. A global estimate for the technical potential of cropland C sequestration is 1.9 (range 0.4–6.8) Gt CO_2-eq yr^{-1}, although socio-economic barriers would probably limit actual C sequestration to only 0.6 (range 0.4–0.9) Gt CO_2-eq yr^{-1} at market prices of US\$100 t^{-1} CO_2-eq (ref. 17). For example, farmers may not adopt technologies owing to their reluctance to making changes, their adversity to taking financial risks and their lack of good information[105,106].

Most agricultural soils have a C-to-N (C:N) ratio of about 12 (ref. 107); thus fixing C in agricultural soil will inevitably impact N cycles in agricultural land[108,109] (Fig. 3d). For example, sequestering 0.6 Gt CO_2-eq yr^{-1} in agricultural soils would result in 14 Tg N yr^{-1} also being sequestered[110] (Table 1; Supplementary Fig. 5), which is equivalent to approximately 12% of current global fertilizer consumption. However, the actual amount of N sequestered in the soil associated with C will vary depending on the C:N ratio of the sequestered forms of soil organic matter and the sources of the N. This important role of N in C sequestration does not necessarily mean that more fertilizer N is needed, because there are opportunities to improve soil retention of current applications of manure and fertilizer N (ref. 58). Hence, the N requirements of newly sequestered soil C also depend on the historical intensity of N use and the management practices that affect N retention or loss.

In intensively managed croplands, particularly those in the developed world, new sequestration of soil C and N could be achieved without additional N application by adopting more sustainable management practices. Such practices can increase crop uptake of N, reduce N losses, increase C inputs to the soil, stabilize soil organic matter and reduce erosion[58]. Winter cover crops and conversion of conventional tillage to no till or reduced tillage have been shown to increase soil C without changing N inputs[111–113]. When fertilizer N is immobilized in the soil through these best management practices, the surplus inorganic-N available to nitrifying and denitrifying bacteria to produce N_2O decreases[114]. Once soil organic matter accumulates, the processes of N mineralization, immobilization and turnover can supply more N for crop needs and N fertilizer inputs and the associated GHG emissions can be reduced[113,115] (Supplementary Fig. 6). Other practices include adding biochar to agricultural soils[111,116] and N-fixing legumes to crop rotations and pastures[61,62]. However, biochar additions can initially immobilize soil N through adsorption of N and reduce N availability in soils, which can result in additional N inputs being required[116]. Biochar applications have been shown to produce medium-term gains in soils C, but important concerns remain about eventual saturation of the soil C sink, permanence of sequestration of the soil C, and viability of C market incentives that require measurement, reporting and validation[106,117]. In addition, production of biochar requires removing biomass stock from another ecosystem, which could affect C and N balances there[116]. Hence, the short-term and long-term impacts of biochar addition on NUE and the N cycle are unclear.

In contrast to intensively managed cropland in the developed world, croplands in many developing countries produce very low yields and at the cost of mining soil N — removing more N in the harvest than the sum of all inputs of N to the soil, thus depleting soil N, and reducing soil C[16,118]. Consequently, soil C sequestration in those croplands probably requires more N inputs, which would increase crop yield, increase crop residue inputs to the soil, and increase soil C and N stocks, with only modest effects on N_2O emissions and N loss through leaching[119]. For example, in a 2-year experiment in Kenya, annual N_2O emissions remained relatively low despite increasing fertilizer application rates in maize production from 0 to 50 kg ha^{-1} yr^{-1} (ref. 119).

The climate benefits of cropland soil C sequestration are determined by the amount of C sequestered and the GHG emissions associated with required N inputs. For example, if increased N inputs owing to C sequestration initiatives are derived from fertilizers produced by the Haber–Bosch process, then the GHG benefit of 1 kg soil C sequestration is reduced by about 7% owing to the embedded CO_2 emissions in the N fertilizer. N_2O emission associated with the increasing N inputs will further offset the C sequestration benefit[108]. Although the increase in N_2O emissions is relatively small in croplands with low-to-moderate N fertilizer inputs, it can increase considerably in heavily fertilized soils owing to the nonlinear response of N_2O emissions to N application rates[120].

In summary, whether C sequestration in agricultural soils yields a net climate benefit depends upon the management of soil N. Where N additions are already small, as in most of sub-Saharan Africa, increasing fertilization rates will probably increase both yields and soil C, but where N application rates are already high, as in North America, Europe and East Asia, soil C sequestration would only have a net climate benefit if current N applications are better managed and increased N application is avoided.

Intensifying crop yields to reduce land use change

Land use change has resulted in net CO_2 emissions of 4.5 ± 2.6 Gt CO_2 yr^{-1} globally during the decade 2012–2021 (ref. 102). These emissions have been predominantly owing to deforestation, which contributed 6.6 ± 1.5 Gt CO_2 yr^{-1} (ref. 102). This value is higher than net emissions from land use change because land use change estimates are the sum of emissions from deforestation (positive flux) and sequestrated CO_2 from activities such as afforestation and reforestation (negative flux)[102]. Deforestation is primarily driven by expansion of agricultural production[17], which subsequently impacts the N cycle. Most agriculture-driven deforestation and its associated CO_2 emissions currently occur in developing countries, especially those in tropical regions[121]. For tropical forests, the median C densities of primary vegetation and undisturbed soils are 190 and 120 t C ha^{-1}, respectively[122]. The conversion of tropical forests to cropland results in a global average C loss of 120 t C ha^{-1}, but the actual loss varies widely depending on factors such as the disturbance to soil C, usage of the woody products, crop type and management practices[123]. It is estimated that collectively Brazil, Indonesia and the Democratic Republic of the Congo alone accounted for approximately 58% of the global emission from land use change between 2012 and 2021 (ref. 102). Thus, halting deforestation in these carbon-dense regions is vital for global climate mitigation, as well as biodiversity conservation. However, stopping tropical deforestation requires global cooperation because the agriculture expansion in these countries is driven by not only their growing domestic needs but also the food, fuel and fibre demand of other countries[124].

One way to slow down agricultural expansion is through intensification[54] whereby the productivity of existing agricultural land is enhanced, which will probably require additional N inputs in many parts of the world (DS5)[125] (Fig. 3e; Supplementary Fig. 7). Taking major cereal production as an example, many parts of the world still have yields that are substantially lower than 75% of the attainable yield, with N input being a major limiting factor for regions such as sub-Saharan Africa and Eastern Europe[126]. Bringing yields up to 75% of the attainable yield would increase global cereal production by 29%, aligning it with the projected increase of 27% to 46% for three major cereal crops by 2050 (ref. 16). However, achieving 75% of the attainable yield would also require a 30% increase in N inputs[126–128]. By achieving 100% of the attainable yield, production for major cereals could increase by 45%–70% (ref. 126), demonstrating the possibility of meeting the 2050 food demand without expanding cropland[129].

The increase in N inputs required to support increases in yield will probably increase N losses to the environment[130]. That said, an initial 50 kg N ha^{-1} yr^{-1} N input rate typically has minimal impacts on N_2O emissions and nitrate leaching[119,131]. Intensification can also be achieved by increasing productivity in pasture and relocating pasture to cropland[132,133]. Both strategies could require more fertilizer N or biological N fixation inputs and are constrained by climate and soil conditions[134]. Thus, when achieving agricultural intensification and increased crop yields, it is crucial to consider how N inputs can best be managed and improved to minimize losses to the environment. Agricultural intensification often also requires additional resources beyond N, such as phosphorus, potassium, lime, other nutrients and irrigation water[127,128], which are beyond the scope of this study but should not be overlooked.

The future efficacy of agricultural intensification remains unclear as questions persist over whether increasing yields will curtail agricultural land expansion or drive further expansion owing to heightened economic incentives[135]. The rebound effect debate[136] is particularly acute for developing countries where agricultural exports are an important component of the economy. The impact of intensification on halting deforestation remains uncertain as it is unclear whether yield increases, especially in the developing tropics, will attract more agricultural activity to these regions[137]. To ensure a favourable outcome, intensification needs to be accompanied by several other measures, such as strong policy intervention on land conversion, effective management of the food supply chains, and international cooperation mechanisms.

Quantifying the impact of agricultural intensification on C emissions attributed to land use is challenging. It is difficult to identify where the potential land savings occur. In the context of an interconnected global market, a yield increase in one location can have complex effects on proximal and distal land conversions. Given the vast disparities in C intensity across natural habitats and climatic zones, identifying where land savings have been achieved is pivotal to quantifying the C benefits of the land-sparing strategy. The global average C stock reduction owing to land conversion from forest to cropland is approximately 120 t C ha^{-1} in the tropics, roughly double that of temperate and boreal zones[123]. There are differences between the C intensities of systems even within the same climatic area, for example, between peat forests and non-peat forests[138]. Thus, verified and standardized protocols and agreements are essential to circumvent double-counting or miscalculating the impacts of avoided land conversions and CO_2 mitigation. Another challenge is accounting for additional N_2O emissions associated with yield increases, which act to reduce the overall climate benefits of the intensification strategy[139]. However, the extent of this reduction is influenced by several factors, such as climate, soil, and management practices.

Overall, intensifying crop yield has large potentials for decarbonization, but its actual impacts on land use change, as well as associated CO_2 emissions and N inputs, warrants further research.

Nitrogen use of other decarbonization strategies

The five decarbonization strategies considered here have varying potentials for CO_2 reduction, from 0.04 to 3.4 Gt CO_2-eq yr^{-1}, but their impact on N inputs can be substantial with some reducing and some increasing N inputs (Fig. 5). Among these strategies, only the strategy of reducing fertilizer application (DS1b) shows a clear co-benefit for CO_2 mitigation and reduction in N inputs, achieving a reduction in N input of 0.36 (0.24–0.41) kg N input per kg CO_2 mitigation. However, DS1b might lead to other issues relating to food security and agricultural practices, such as potentially scaled-down biomass production, supply-chain mismanagement, or strained shifts in farming and food consumption choices[39]. By contrast, using NH_3 as fuel (DS3) will lead to an increase in N inputs, ranging from 0.5 to 14 kg N input per kg CO_2 mitigation. Other strategies might have negligible impacts on N input (for example, DS1a) or could increase the N input at the rate up to 0.08 kg N input per kg CO_2 mitigation (for example, DS5). Large uncertainties remain in these initial assessments; however, the metric of N input change per kg CO_2 mitigation can help to facilitate the discussion and comparison of various decarbonization strategies.

Other decarbonization strategies could also potentially impact eutrophication risk by influencing the fate of N and other nutrients. In the energy sector, clean-energy production from solar, wind, hydropower and geoelectric alternatives help offset CO_2 released from burning fossil fuels but can have a N footprint[140]. For example, large water reservoirs built for hydroelectric power production can alter upstream and downstream N flows[140]. Water impoundment upstream of hydroelectric dams could lead to increased water residence times, elevated nutrient concentrations, eutrophication and hypoxia[140]. Downstream of dams and changes to water streamflow affect the distribution and availability of nutrients. Reductions in riverine nutrients could impact nutrient supply to the farmlands, which could then impact agricultural productivity and the feasibility of producing bioenergy crops as an alternative energy source to fossil fuels[140]. By comparison, reducing the use of fossil fuels in the energy sector, an important component

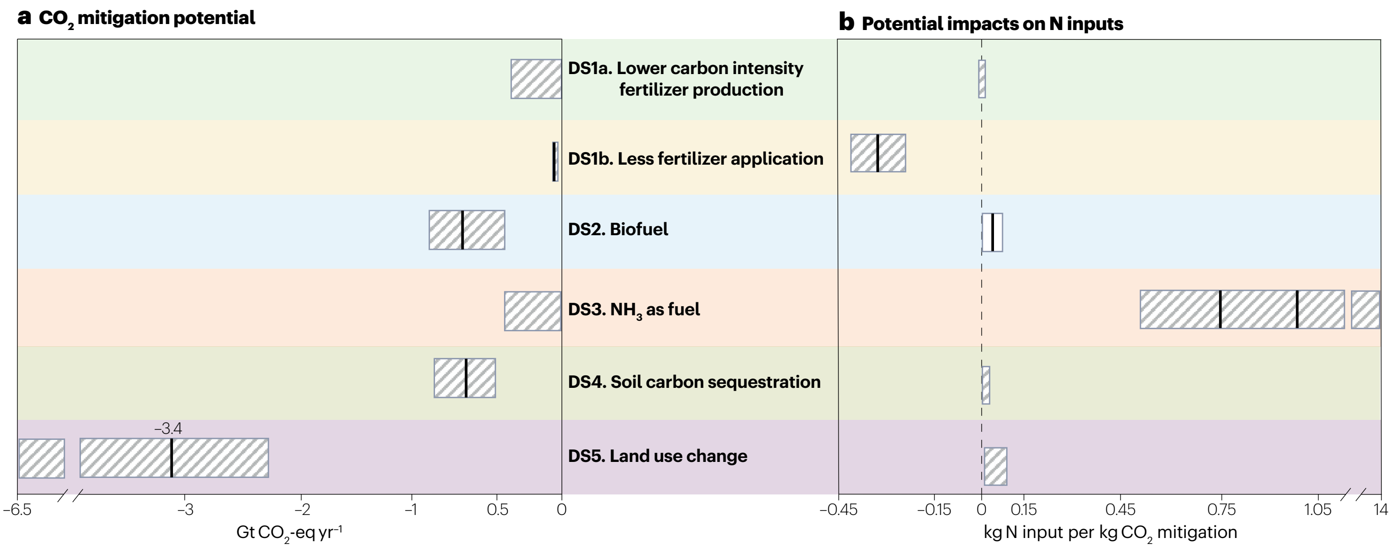

Fig. 5 | CO_2 mitigation potential of each decarbonization strategy and the impact on N inputs per kg CO_2 reduction. a, CO_2 mitigation potential of each decarbonization strategy. **b**, Potential impacts on N inputs of each decarbonization strategy. Boxes with grey lines represent the uncertainty range of CO_2 mitigation potential (Gt CO_2-eq yr^{-1}) and N impacts (kg N input per kg CO_2 mitigation), and the black lines in the boxes represent the mean values. Negative values for N impacts indicate the decrease in N inputs per unit of CO_2 mitigation. The two black lines in panel **b** for DS3 correspond to average values for NH_3 produced from solar (1.02 kg N input per kg CO_2 mitigation) and wind (0.75 kg N input per kg CO_2 mitigation) energy. Detailed information could be found in Supplementary Note 3. Different decarbonization strategies have varying impacts on CO_2 emission and N inputs.

of decarbonization, will have co-benefits of reducing NO_x and N_2O emissions to the atmosphere and consequently reduce human-induced perturbations of the N cycle[141].

Nutrients such as phosphorus and potassium are also affected by decarbonization strategies. In the energy sector, phosphorus and potassium are important in the development of advanced battery systems for energy storage[142]. For instance, compounds containing phosphorus, such as lithium iron phosphate, and emerging potassium-ion batteries as alternatives to lithium-ion ones have important roles in renewable energy infrastructures, and their availability influences the electrification potential in end-use sectors, such as the transport sector[143]. However, comprehensively assessing all the various decarbonization endeavours and their influence on all nutrient cycles is beyond the scope of this article.

Links between decarbonization strategies

The five decarbonization strategies considered here are interconnected, as they all relate to N use in agricultural production, indicating a need for better coordination among various decarbonization efforts. This section summarizes the dynamics and potential synergistic and antagonistic effects of these decarbonization strategies and highlights the need for effective coordination between the various decarbonization efforts to mitigate excessive N use.

Yield response to N input on crop farms

For a given area of cropland, the crop yield is linked to N inputs and their management[56]. In DS5, the aim is to achieve higher yield per unit area of cropland. However, depending on how this higher yield is achieved it could impact other decarbonization strategies associated with N management in croplands, for example, DS1 and DS4. The combined additional N inputs needed by DS4 and DS5 are larger than, or similar to, the potential reduction by DS1 (Table 1), suggesting that progress made by DS1 in reducing N inputs could be overshadowed by the increasing N demands by DS4 and DS5.

Yield responses to N inputs for a given cropland and management practice are often represented with yield response curves, which show a diminishing return of yield increase as N input increases[56] (Supplementary Fig. 8). Yield increase can be achieved by increasing N input within an established practice to intensify production along the same yield curve or, alternatively, by adopting a new practice that shifts production from one yield curve to another[56]. The potential yield gain from the first approach is limited if the existing cropland management practice is already operating close to the maximum achievable yield, meaning that progressively greater N inputs will be required to achieve further yield gains[117]. Intensive increases in N inputs such as these will counteract DS1, which seeks to reduce carbon-intensive fertilizer use. By contrast, the second approach largely aligns with the management goals of DS1 and DS4, as it achieves higher yield through improved N management and higher N use efficiency. But available N management practices that can reduce N input and enhance C sequestration, such as cover crop and no-till farming, might not achieve yield levels required by DS5 to avoid additional land conversion[113]. The overlapping opportunities for DS1 and DS5 are in regions with low N input and yield, but it is possible that some additional N inputs are needed to build up soil C and N stocks (DS4) at the initial stages of efforts to increase yield[144].

Competing demand for crops

Although it is widely accepted that crop production must increase to feed the growing global population[69,145], crops are also increasingly being used for purposes beyond satisfying basic protein and caloric needs for humans alone. In the case of US corn, approximately 40% of corn is used for ethanol, 40% is used for domestic animal feed, and

20% is used for direct human consumption, seeds, other industrial uses, and exports[72]. Of the latter 20%, exports make up almost 40% (see US Department of Agriculture Economic Research Service) and, as these exports are mostly used for animal feed, only a small fraction remains for direct human consumption[41] (Fig. 4). Therefore, the increasing demand to use crops as fuel (DS2) and feed is probably going to counteract the goal of reducing N fertilizer use in crop production (DS1) and elevate the pressure for land conversion to more intensive crop production (DS5).

The impacts of DS2 on other decarbonization strategies can potentially be alleviated by effective recycling and utilization of N in crop, animal, and energy production systems[33,83,146]. Since 2012 in the USA, between 80% to 95% of N in corn sent to the biorefinery is recovered in dried distillers grains that is used for livestock feed (see US Department of Agriculture). Conversely, only 35% of livestock manure is recycled back to cropland, leaving the remaining N contained in manure to be lost into the environment. Ineffective manure recycling not only causes a wide range of environmental problems but also wastes useful resources[110]. The pressure for additional N inputs for feed and fuel production can be alleviated through improved management of the cycling of N between crop, animal and biofuel production. For example, additional inputs of 21 ± 9 Tg N yr^{-1} needed for producing corn ethanol to meet the mitigation potential could be met by recycling currently wasted manure[147] (Table 1).

Balancing market dynamics

Efficiency gains in agriculture–food–fuel production will interact with market demand, subsequently influencing the overall effectiveness of these strategies in reducing CO_2 emissions and N pollution. For instance, enhancing NUE and yield in crop production could allow for the same amount of fuel and food to be produced with fewer N inputs and less land (DS1 and DS5). However, it might also spur higher demands for biofuel (DS2) and food or shift the demands toward more N-intensive, land-intensive or water-intensive food and fuel products[16,148]. To avoid potential rebound effects, governance structures are needed to limit conversion of lands to agriculture and unsustainable withdrawals of water from surface and groundwater sources[149].

Another important market dynamic is associated with the growing need for NH_3 as fuel (DS3), as well as the disruption of the fertilizer supply chain caused by the distributed and low-C fertilizer production system (DS1a). These changes tend to affect the market price and supply of fertilizer N and consequently influence the management of N throughout the agriculture–food–energy system (DS1b and DS3), although the actual impacts, and even the sign of the impacts, are still largely unknown[52].

Summary and future perspectives

Decarbonization strategies have complex impacts on the global N cycle. Overlooking N use when addressing CO_2 emissions could undermine C-based climate mitigation benefits and lead to other environmental issues, such as eutrophication. Some decarbonization strategies, such as DS1 which reduces C-intensive synthetic N fertilizer use, could reduce N pollution and offer co-benefits of mitigating climate change and eutrophication (Fig. 5). However, several other strategies, including DS2 and DS3 that use crop-based biofuel and NH_3 as fuels, have substantial N_r demands that will impact the global N cycle and food security[98]. Among the five strategies considered here, using NH_3 as a fuel will probably have the largest impact on human N use (Fig. 5). Therefore, given the growing emphasis on decarbonization in private and public sectors, it is imperative to elevate efforts to address eutrophication concerns alongside climate change.

Technological advancements in agricultural N management can help simultaneously mitigate climate change and eutrophication. Farm-level practices and technologies to boost NUE range from existing methods such as conservation tillage and cover crops to cutting-edge approaches involving gene editing and artificial intelligence[150]. Broader systemic changes that improve N management beyond farms remain a pressing necessity[151,152]. In the current agriculture–food system, only 16% of agricultural N inputs end up in human food consumption[24] despite 46% NUE for global cropland[153], which highlights the inefficiency of C-intensive fertilizer products and high N losses from livestock and supply chains. Using technologies that promote recycling of manure and food waste, prevent food loss and waste, and ensure efficient food delivery from farm to table can reduce N loss from supply chains[31]. For example, acidifying manure and adding compounds, such as aluminum sulfate, biochar, and clay, to manure could help adjust N-to-phosphorous ratios through reducing NH_3 emissions and/or phosphorus leaching to better meet crop needs[154]. In addition, thermal processing technologies could improve the conversion of human food waste into animal feed, which reduces N loss from food waste and the N input requirements of producing feed grains, thereby further mitigating N loss[155].

Technological innovation alone cannot resolve climate change and eutrophication. Socioeconomic innovations that foster policy and market conditions favourable to technology development, implementation and adoption are also needed. Such policy and market instruments include subsidies for cover crops[156], crop insurance programmes, and NUE mandates by food retailers[152]. Policies at both national and international scales can establish safeguards against unintended consequences. For instance, the intensification strategy (DS5) is viable only with robust land governance, such as the Roundtable on Sustainable Palm Oil (see Roundtable on Sustainable Palm Oil) and the REDD+ programme (see United Nations Framework Convention on Climate Change). Without land governance, increased yields and profitability from deforested lands could further entice exploitation of regional forest resources. Finally, innovations in market strategies and education can steer consumption behaviours away from products that result in intensive GHG emissions and N pollution, such as moving toward more plant-based diets.

Managing climate change and eutrophication challenges together requires a solid grasp of how C and N cycles interact across human and natural systems. Advances in Earth system modelling of biogeochemical cycles have provided insights into links between C and N cycles in and across terrestrial and aquatic ecosystems, including the potential impacts of decarbonization strategies. State-of-the-art crop models operating across multiple spatial scales can support more efficient N management on farms[157,158] and enable better assessment of N emissions and environmental outcomes[139,159,160]. Advances in observational capacities and datasets, from point measurement of N and C concentrations in soil and water to regional atmospheric measurements and remote sensing, have provided valuable insights into how decarbonization strategies will probably affect demand for and fates of N.

Quantifying and monitoring N flows in human systems, particularly agriculture–food–energy systems, remain limited[161,162]. Advances in tracking N across multiple human systems on national to global scales[161,163] have enabled Integrated Assessment Models to project potential impacts of socioeconomic conditions and technological development on human N use and GHG emissions[164,165]. For example, the Global Change Analysis Model was used to project the global and regional consequences up to 2100 of realizing the mitigation of

GHG emissions, shifting to predominately plant-based diet, and yield intensification[165]. However, challenges remain, including limitations in our understanding and representation of intricate N flows in the food supply chain[166]. Data availability is an issue when assessing N flows through food supply chains, especially for the processing and retail stages[24,167]. Additionally, interregional and international trade complicate the definition of system boundaries when conducting N budget assessments, as trade and recycling occur across different spatial scales. Owing to privacy and regulatory concerns, there is also a lack of basic data such as farm-level N application and information on management practices[153]. Developing secure digital platforms for anonymized reporting with standardized data collection protocols could help improve data availability for N management assessment and help overcome these challenges. Collaborating with farmers to promote the benefits of sharing data and providing economic incentives to encourage participation in data reporting could also help address data limitation issues.

There is ongoing debate about the robustness of some methods used to evaluate the effectiveness of strategies to mitigate climate change and eutrophication[168,169]. For instance, differences between methods for estimating the C intensity of ethanol production, and whether they focus solely on impacts from land use change or consider other aspects, could result in divergent evaluations of biofuel use. Increasing the circularity of N and C in the agriculture–food–energy system and enhancing material use efficiencies, such as reusing ethanol by-products as feed, will bring additional challenges in C and N accounting[170,171]. Estimates suggest that in 2010, 2% of global cropland was dedicated to biofuel production, declining to 1.5% when the use of biofuel co-products as livestock feed is considered, owing to the reduction in land area required for cultivating grains and oilseeds[170]. Omitting co-product generation in biofuel assessments could, therefore, lead to land requirements being overestimated by over 40% (ref. 170). In the future, models assessing biofuel policies should integrate impacts of co-products on land use, economy and environment.

Minimizing the impacts of decarbonization on the global N cycle requires that technology, socioeconomic and systemic changes across the agriculture–food–energy system converge on the aim of addressing climate change and nutrient pollution together. It is critical to account for and manage the N dimensions of decarbonization strategies to ensure food security and minimize environmental degradation as we mitigate CO_2 emissions. This multifaceted challenge underscores the importance of a holistic, integrated perspective when designing and implementing sustainable solutions, ensuring that our pursuit of a carbon-neutral future does not inadvertently exacerbate other environmental challenges.

Data availability

All data are available in the article and its Supplementary information. The data and associated R scripts for Fig. 1 are openly available in Zenodo at https://zenodo.org/records/11097234 (ref. 176). The data for Fig. 5 is available in Supplementary Table 1.

References

1. Rockström, J. et al. Safe and just Earth system boundaries. *Nature* **619**, 102–111 (2023).
2. Richardson, K. et al. Earth beyond six of nine planetary boundaries. *Sci. Adv*. **9**, eadh2458 (2023).
3. Steffen, W. et al. Planetary boundaries: guiding human development on a changing planet. *Science* https://doi.org/10.1126/science.1259855 (2015).
4. IPCC. *Climate Change 2021*: *The Physical Science Basis*. *Summary for Policymakers Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Masson-Delmotte, V. et al.) (Cambridge Univ. Press, 2021).
5. Dhakal, S. et al. in *Climate Change 2022*: *Mitigation of Climate Change* (eds Shukla, P. R. et al.) Ch. 2 (Cambridge Univ. Press, 2022).
6. McKay, D. I. A. et al. Exceeding 1.5 °C global warming could trigger multiple climate tipping points. *Science* **377**, eabn7950 (2022).
7. Sinha, E., Michalak, A. M. & Balaji, V. Eutrophication will increase during the 21st century as a result of precipitation changes. *Science* **357**, 405–408 (2017).
8. Suddick, E. C., Whitney, P., Townsend, A. R. & Davidson, E. A. The role of nitrogen in climate change and the impacts of nitrogen-climate interactions in the United States: foreword to thematic issue. *Biogeochemistry* https://doi.org/10.1007/s10533-012-9795-z (2013).
9. Gruber, N. & Galloway, J. N. An Earth-system perspective of the global nitrogen cycle. *Nature* **451**, 293–296 (2008).
10. Wolfram, P., Kyle, P., Zhang, X., Gkantonas, S. & Smith, S. Using ammonia as a shipping fuel could disturb the nitrogen cycle. *Nat. Energy* **7**, 1112–1114 (2022).
11. Glibert, P. M. Eutrophication, harmful algae and biodiversity — challenging paradigms in a world of complex nutrient changes. *Mar. Pollut. Bull.* **124**, 591–606 (2017).
12. Ho, J. C., Michalak, A. M. & Pahlevan, N. Widespread global increase in intense lake phytoplankton blooms since the 1980s. *Nature* **574**, 667–670 (2019).
13. Breitburg, D. et al. Declining oxygen in the global ocean and coastal waters. *Science* **359**, eaam7240 (2018).
14. de Raús Maúre, E., Terauchi, G., Ishizaka, J., Clinton, N. & DeWitt, M. Globally consistent assessment of coastal eutrophication. *Nat. Commun*. https://doi.org/10.1038/s41467-021-26391-9 (2021).
15. Zhou, X. et al. Ammonia marine engine design for enhanced efficiency and reduced greenhouse gas emissions. *Nat. Commun*. https://doi.org/10.1038/s41467-024-46452-z (2024).
16. Zhang, X. et al. Managing nitrogen for sustainable development. *Nature* **528**, 51–59 (2015).
17. Nabuurs, G.-J. et al. in *Climate Change 2022*: *Mitigation of Climate Change* (eds Shukla, P. R. et al.) Ch. 7 (Cambridge Univ. Press, 2022).
18. Tian, H. et al. A comprehensive quantification of global nitrous oxide sources and sinks. *Nature* **586**, 248–256 (2020).
19. Liang, X. et al. Air quality and health benefits from fleet electrification in China. *Nat. Sustain.* **2**, 962–971 (2019).
20. Galloway, J. N. et al. The nitrogen cascade. *Bioscience* **53**, 341–356 (2003).
21. Erisman, J. W., Sutton, M. A., Galloway, J., Klimont, Z. & Winiwarter, W. How a century of ammonia synthesis changed the world. *Nat. Geosci.* **1**, 636–639 (2008).
22. David, F. et al. The global nitrogen cycle in the twenty-first century. *Philos. Trans. R. Soc. Lond. B Biol. Sci.* **368**, 20130164 (2013).
23. Ludemann, C. I. et al. A global FAOSTAT reference database of cropland nutrient budgets and nutrient use efficiency (1961-2020): nitrogen, phosphorus and potassium. *Earth Syst. Sci. Data* https://doi.org/10.5194/essd-16-525-2024 (2024).
24. Zhang, X. et al. Quantifying nutrient budgets for sustainable nutrient management. *Glob. Biogeochem. Cycles* https://doi.org/10.1029/2018GB006060 (2020).
25. Schulte-Uebbing, L. F., Beusen, A. H. W., Bouwman, A. F. & de Vries, W. From planetary to regional boundaries for agricultural nitrogen pollution. *Nature* **610**, 507–512 (2022).
26. Guo, Y. et al. Environmental and human health trade-offs in potential Chinese dietary shifts. *One Earth* **5**, 268–282 (2022).
27. Scientific Panel on Responsible Plant Nutrition. Achieving nature-positive plant nutrition: fertilizers and biodiversity. *SPRPN Issue Brief* Issue Brief 02 (2021).
28. Bobbink, R. et al. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecol. Appl.* **20**, 30–59 (2010).
29. Kanter, D. R. et al. Improving the social cost of nitrous oxide. *Nat. Clim. Change* **11**, 1008–1010 (2021).
30. Billen, G., Lassaletta, L. & Garnier, J. A biogeochemical view of the global agro-food system: nitrogen flows associated with protein production, consumption and trade. *Glob. Food Secur.* **3**, 209–219 (2014).
31. Marconi, P. & Rosa, L. Global potential nitrogen recovery from anaerobic digestion of agricultural residues. *Environ. Res. Lett.* **19**, 054050 (2024).
32. Bai, Z. et al. Relocate 10 billion livestock to reduce harmful nitrogen pollution exposure for 90% of China's population. *Nat. Food* **3**, 152–160 (2022).
33. Spiegal, S. et al. Manuresheds: advancing nutrient recycling in US agriculture. *Agric. Syst.* **182**, 102813 (2020).
34. Sabo, R. D., Clark, C. M. & Compton, J. E. Considerations when using nutrient inventories to prioritize water quality improvement efforts across the us. *Environ. Res. Commun.* **3**, 045005 (2021).
35. Sabo, R. D. et al. Decadal shift in nitrogen inputs and fluxes across the contiguous United States: 2002–2012. *J. Geophys. Res. Biogeosci*. https://doi.org/10.1029/2019JG005110 (2019).
36. International Energy Agency. Ammonia technology roadmap (IEA, 2021).
37. Gao, Y. & Cabrera Serrenho, A. Greenhouse gas emissions from nitrogen fertilizers could be reduced by up to one-fifth of current levels by 2050 with combined interventions. *Nat. Food* **4**, 170–178 (2023).
38. Menegat, S., Ledo, A. & Tirado, R. Greenhouse gas emissions from global production and use of nitrogen synthetic fertilisers in agriculture. *Sci. Rep.* **12**, 14490 (2022).

39. Rosa, L. & Gabrielli, P. Energy and food security implications of transitioning synthetic nitrogen fertilizers to net-zero emissions. *Environ. Res. Lett.* **18**, 014008 (2023).
40. Lange, J. P. Towards circular carbo-chemicals-the metamorphosis of petrochemicals. *Energy Environ. Sci*. https://doi.org/10.1039/D1EE00532D (2021).
41. Gabrielli, P. et al. Net-zero emissions chemical industry in a world of limited resources. *One Earth* **6**, 682–704 (2023).
42. Rosa, L. & Gabrielli, P. Achieving net-zero emissions in agriculture: a review. *Environ. Res. Lett.* **18**, 063002 (2023).
43. Tonelli, D. et al. Global land and water limits to electrolytic hydrogen production using wind and solar resources. *Nat. Commun.* **14**, 5532 (2023).
44. Arnaiz del Pozo, C. & Cloete, S. Techno-economic assessment of blue and green ammonia as energy carriers in a low-carbon future. *Energy Convers. Manag.* **255**, 115312 (2022).
45. Cardoso, J. S. et al. Ammonia as an energy vector: current and future prospects for low-carbon fuel applications in internal combustion engines. *J. Clean Prod.* **296**, 126562 (2021).
46. Chehade, G. & Dincer, I. Progress in green ammonia production as potential carbon-free fuel. *Fuel* **299**, 120845 (2021).
47. Salmon, N. & Bañares-Alcántara, R. Green ammonia as a spatial energy vector: a review. *Sustain. Energy Fuels* https://doi.org/10.1039/D1SE00345C (2021).
48. Lee, B. et al. Pathways to a green ammonia future. *ACS Energy Lett*. https://doi.org/10.1021/acsenergylett.2c01615 (2022).
49. Ashida, Y., Arashiba, K., Nakajima, K. & Nishibayashi, Y. Molybdenum-catalysed ammonia production with samarium diiodide and alcohols or water. *Nature* **568**, 536–540 (2019).
50. Ashida, Y. et al. Catalytic nitrogen fixation using visible light energy. *Nat. Commun.* **13**, 7263 (2022).
51. Fu, X. et al. Continuous-flow electrosynthesis of ammonia by nitrogen reduction and hydrogen oxidation. *Science* **379**, 707–712 (2023).
52. Tonelli, D., Rosa, L., Gabrielli, P., Parente, A. & Contino, F. Cost-competitive decentralized ammonia fertilizer production can increase food security. *Nat. Food* **5**, 469–479 (2024).
53. Davidson, E. A. et al. Nutrients in the nexus. *J. Environ. Stud. Sci*. **6**, 25–38 (2016).
54. Burney, J. A., Davis, S. J. & Lobell, D. B. Greenhouse gas mitigation by agricultural intensification. *Proc. Natl Acad. Sci. USA* **107**, 12052–12057 (2010).
55. Cassman, K. G. & Grassini, P. A global perspective on sustainable intensification research. *Nat. Sustain.* **3**, 262–268 (2020).
56. Zhang, X., Mauzerall, D. L., Davidson, E. A., Kanter, D. R. & Cai, R. The economic and environmental consequences of implementing nitrogen-efficient technologies and management practices in agriculture. *J. Env. Qual*. **44**, 312–324 (2015).
57. Li, T. et al. Exploring optimal nitrogen management practices within site-specific ecological and socioeconomic conditions. *J. Clean Prod.* **241**, 118295 (2019).
58. Quan, Z. et al. Fates and use efficiency of nitrogen fertilizer in maize cropping systems and their responses to technologies and management practices: a global analysis on field 15N tracer studies. *Earths Future* https://doi.org/10.1029/2020EF001514 (2021).
59. Cai, S. et al. Optimal nitrogen rate strategy for sustainable rice production in China. *Nature* **615**, 73–79 (2023).
60. Cui, X. et al. The global potential for mitigating nitrous oxide emissions from croplands. *One Earth* **7**, 401–420 (2024).
61. Zhao, S. et al. A precision compost strategy aligning composts and application methods with target crops and growth environments can increase global food production. *Nat. Food* **3**, 741–752 (2022).
62. Bhattacharjee, R. B., Singh, A. & Mukhopadhyay, S. N. Use of nitrogen-fixing bacteria as biofertiliser for non-legumes: prospects and challenges. *Appl. Microbiol. Biotechnol.* **80**, 199–209 (2008).
63. Mus, F. et al. Symbiotic nitrogen fixation and the challenges to its extension to nonlegumes. *Appl. Environ. Microbiol.* **82**, 3698–3710 (2016).
64. Yan, D. et al. Genetic modification of flavone biosynthesis in rice enhances biofilm formation of soil diazotrophic bacteria and biological nitrogen fixation. *Plant Biotechnol. J.* **20**, 2135–2148 (2022).
65. Chadwick, D. et al. Manure management: implications for greenhouse gas emissions. *Anim. Feed Sci. Technol.* **166–167**, 514–531 (2011).
66. Zhang, X. & Lassaletta, L. Manure management benefits climate with limits. *Nat. Food* **3**, 312–313 (2022).
67. Basso, B., Jones, J. W., Antle, J., Martinez-Feria, R. A. & Verma, B. Enabling circularity in grain production systems with novel technologies and policy. *Agric. Syst.* **193**, 103244 (2021).
68. Billen, G. et al. Reshaping the European agro-food system and closing its nitrogen cycle: the potential of combining dietary change, agroecology, and circularity. *One Earth* **4**, 839–850 (2021).
69. Gustavsson, J., Cederberg, C., Sonesson, U., Van Otterdijk, R. & Meybeck, A. *Global Food Losses and Food Waste: Extent, Causes and Prevention* (FAO, 2011).
70. Gustavsson, J., Cederberg, C., Sonesson, U. & Emanuelsson A. The methodology of the FAO study: "Global Food Losses and Food Waste — extent, causes and prevention" — FAO, 2011 (SIK, 2013).
71. Chatzimpiros, P. & Harchaoui, S. Sevenfold variation in global feeding capacity depends on diets, land use and nitrogen management. *Nat. Food* **4**, 372–383 (2023).
72. Liu, Y. et al. Biofuels for a sustainable future. *Cell* **184**, 1636–1647 (2021).
73. Lark, T. J. et al. Environmental outcomes of the US renewable fuel standard. *Proc. Natl Acad. Sci. USA* **119**, e2101084119 (2022).
74. Scully, M. J., Norris, G. A., Alarcon Falconi, T. M. & MacIntosh, D. L. Carbon intensity of corn ethanol in the United States: state of the science. *Environ. Res. Lett.* **16**, 043001 (2021).
75. Lee, R. A. & Lavoie, J. M. From first- to third-generation biofuels: challenges of producing a commodity from a biomass of increasing complexity. *Anim. Front.* **3**, 6–11 (2013).
76. Crutzen, P. J., Mosier, A. R., Smith, K. A. & Winiwarter, W. N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmos. Chem. Phys.* **8**, 389–395 (2008).
77. Delucchi, M. A. Impacts of biofuels on climate change, water use, and land use. *Ann. N. Y. Acad. Sci.* **1195**, 28–45 (2010).
78. Jeswani, H. K., Chilvers, A. & Azapagic, A. Environmental sustainability of biofuels: a review: environmental sustainability of biofuels. *Proc. Math. Phys. Eng. Sci.* **476**, 20200351 (2020).
79. US Environmental Protection Agency. Renewable Fuel Standard Program (RFS2) regulatory impact analysis (USEPA, 2010).
80. US Environmental Protection Agency. Model comparison exercise technical document (USEPA, 2023).
81. Shalaby, E. A. in *Liquid, Gaseous and Solid Biofuels—Conversion Techniques* (ed. Fang, Z.) (IntechOpen, 2013).
82. Shurson, G. C., Tilstra, H. & Kerr, B. J. in *Biofuel Co-products as Livestock Feed — Opportunities and Challenges* Ch. 3 (FAO, 2012).
83. Algren, M., Landis, A. E. & Costello, C. Estimating virtual nitrogen inputs to integrated U.S. corn ethanol and animal food systems. *Environ. Sci. Technol.* **55**, 8393–8400 (2021).
84. Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J. & Garnier, J. 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. *Environ. Res. Lett.* **9**, 105011 (2014).
85. Ates, A. M. & Liefert, O. Feed outlook: October 2023 (report no. FDS-23j) (USDA, 2023).
86. Nanda, S., Rana, R., Sarangi, P. K., Dalai, A. K. & Kozinski, J. A. in *Recent Advancements Biofuels Bioenergy Utilization* Ch. 1 (Springer, 2018).
87. Dragone, G., Fernandes, B., Vicente, A. & Teixeira, J. in *Current Research, Technology and Education Topics in Applied Microbiology and Microbial Biotechnology* Ch. 2 (Formatex Research Center, 2010).
88. Bhuneshwar & Vaishnav, A. The third-generation biofuel production from algae. *AGRIALLIS* **3**, AL202190 (2021).
89. Nguyen, L. N. et al. Nutrient removal by algae-based wastewater treatment. *Curr. Pollut. Rep.* https://doi.org/10.1007/s40726-022-00230-x (2022).
90. Sacchi, R. et al. How to make climate-neutral aviation fly. *Nat. Commun.* **14**, 3989 (2023).
91. Nishina, K. New ammonia demand: ammonia fuel as a decarbonization tool and a new source of reactive nitrogen. *Environ. Res. Lett*. https://doi.org/10.1088/1748-9326/ac4b74 (2022).
92. International Energy Agency. Net zero roadmap: a global pathway to keep the 1.5 °C goal in reach (IEA, 2023).
93. Stolz, B., Held, M., Georges, G. & Boulouchos, K. Techno-economic analysis of renewable fuels for ships carrying bulk cargo in Europe. *Nat. Energy* **7**, 203–212 (2022).
94. Raucci C., McKinlay C. & Karan A. The future of maritime fuels: what you need to know (Lloyd's Register, 2023).
95. Al-Breiki, M. & Bicer, Y. Technical assessment of liquefied natural gas, ammonia and methanol for overseas energy transport based on energy and exergy analyses. *Int. J. Hydrog. Energy* **45**, 34927–34937 (2020).
96. Hauglustaine, D. et al. Climate benefit of a future hydrogen economy. *Commun. Earth Environ*. https://doi.org/10.1038/s43247-022-00626-z (2022).
97. Van Damme, M. et al. Industrial and agricultural ammonia point sources exposed. *Nature* **564**, 99–103 (2018).
98. Bertagni, M. B. et al. Minimizing the impacts of the ammonia economy on the nitrogen cycle and climate. *Proc. Natl Acad. Sci. USA* **120**, e2311728120 (2023).
99. International Renewable Energy Agency & Ammonia Energy Association. Innovation outlook: renewable ammonia (IRENA-AEA, 2022).
100. Glenk, G., Holler, P. & Reichelstein, S. Advances in power-to-gas technologies: cost and conversion efficiency. *SSRN Electron. J.* https://doi.org/10.2139/ssrn.4300331 (2022).
101. Minasny, B. et al. Soil carbon 4 per mille. *Geoderma* **292**, 59–86 (2017).
102. Friedlingstein, P. et al. Global carbon budget 2022. *Earth Syst. Sci. Data* **14**, 4811–4900 (2022).
103. Sanderman, J., Hengl, T. & Fiske, G. J. Soil carbon debt of 12,000 years of human land use. *Proc. Natl Acad. Sci. USA* **114**, 9575–9580 (2017).
104. Zheng, F. et al. Linking soil microbial community traits and organic carbon accumulation rate under long-term conservation tillage practices. *Soil Tillage Res.* **220**, 105360 (2022).
105. Oldfield, E. E., Lavallee, J. M., Kyker-Snowman, E. & Sanderman, J. The need for knowledge transfer and communication among stakeholders in the voluntary carbon market. *Biogeochemistry* https://doi.org/10.1007/s10533-022-00950-8 (2022).
106. Davidson, E. A. Is the transactional carbon credit tail wagging the virtuous soil organic matter dog? *Biogeochemistry* https://doi.org/10.1007/s10533-022-00969-x (2022).
107. Kopittke, P. M., Dalal, R. C., Finn, D. & Menzies, N. W. Global changes in soil stocks of carbon, nitrogen, phosphorus, and sulphur as influenced by long-term agricultural production. *Glob. Chang Biol.* **23**, 2509–2519 (2017).
108. Schlesinger, W. H. Biogeochemical constraints on climate change mitigation through regenerative farming. *Biogeochemistry* **161**, 9–17 (2022).
109. Van Groenigen, J. W. et al. Sequestering soil organic carbon: a nitrogen dilemma. *Environ. Sci. Technol*. https://doi.org/10.1021/acs.est.7b01427 (2017).

110. FAO. *Food and Agriculture Data* (FAOSTAT Online Database, accessed October 2023); https://www.fao.org/faostat/en/#data/EMN.
111. Bossio, D. A. et al. The role of soil carbon in natural climate solutions. *Nat. Sustain.* **3**, 391–398 (2020).
112. Robertson, G. P., Paul, E. A. & Harwood, R. R. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. *Science* **289**, 1922–1925 (2000).
113. Via, S. Increasing soil health and sequestering carbon in agricultural soils: a natural climate solution (Izaak Walton League of America and National Wildlife Federation, 2021).
114. Davidson, E. A. & Firestone, M. K. *Microbiological Basis of NO and N_2O Production and Consumption in Soil* 7–21 (Wiley, 1989).
115. Powlson, D. S. & Galdos, M. V. Challenging claimed benefits of soil carbon sequestration for mitigating climate change and increasing crop yields: heresy or sober realism? *Glob. Chang Biol.* **29**, 2381–2383 (2023).
116. Lehmann, J. et al. Biochar in climate change mitigation. *Nat. Geosci.* **14**, 883–892 (2021).
117. Moinet, G. Y. K., Hijbeek, R., van Vuuren, D. P. & Giller, K. E. Carbon for soils, not soils for carbon. *Glob. Chang. Biol.* **29**, 2384–2398 (2023).
118. de Blécourt, M., Gröngröft, A., Baumann, S. & Eschenbach, A. Losses in soil organic carbon stocks and soil fertility due to deforestation for low-input agriculture in semi-arid southern Africa. *J. Arid Environ.* **165**, 88–96 (2019).
119. Hickman, J. E., Tully, K. L., Groffman, P. M., Diru, W. & Palm, C. A. A potential tipping point in tropical agriculture: avoiding rapid increases in nitrous oxide fluxes from agricultural intensification in Kenya. *J. Geophys. Res. Biogeosci*. https://doi.org/10.1002/2015JG002913 (2015).
120. Eagle, A. J. et al. Quantifying on-farm nitrous oxide emission reductions in food supply chains. *Earths Future* https://doi.org/10.1002/essoar.10502712.1 (2020).
121. Hong, C. et al. Global and regional drivers of land-use emissions in 1961–2017. *Nature* **589**, 554–561 (2021).
122. Houghton, R. A. & Nassikas, A. A. Negative emissions from stopping deforestation and forest degradation, globally. *Glob. Chang. Biol.* **24**, 350–359 (2018).
123. West, P. C. et al. Trading carbon for food: global comparison of carbon stocks vs. crop yields on agricultural land. *Proc. Natl Acad. Sci. USA* **107**, 19645–19648 (2010).
124. Pendrill, F. et al. Disentangling the numbers behind agriculture-driven tropical deforestation. *Science* **377**, eabm9267 (2022).
125. Foley, J. A. et al. Solutions for a cultivated planet. *Nature* **478**, 337–342 (2011).
126. Mueller, N. D. et al. Closing yield gaps through nutrient and water management. *Nature* **490**, 254–257 (2012).
127. Rosa, L., Chiarelli, D. D., Rulli, M. C., Dell'Angelo, J. & D'Odorico, P. Global agricultural economic water scarcity. *Sci. Adv.* **6**, eaaz6031 (2020).
128. Rosa, L. et al. Closing the yield gap while ensuring water sustainability. *Environ. Res. Lett.* **13**, 104002 (2018).
129. Beltran-Pea, A., Rosa, L. & D'Odorico, P. Global food self-sufficiency in the 21st century under sustainable intensification of agriculture. *Environ. Res. Lett.* **15**, 095004 (2020).
130. Rosa, L. Adapting agriculture to climate change via sustainable irrigation: biophysical potentials and feedbacks. *Environ. Res. Lett.* **17**, 063008 (2022).
131. Russo, T. A., Tully, K., Palm, C. & Neill, C. Leaching losses from Kenyan maize cropland receiving different rates of nitrogen fertilizer. *Nutr. Cycl. Agroecosyst.* **108**, 195–209 (2017).
132. Kreidenweis, U. et al. Pasture intensification is insufficient to relieve pressure on conservation priority areas in open agricultural markets. *Glob. Chang. Biol.* **24**, 3199–3213 (2018).
133. Oenema, O., De Klein, C. & Alfaro, M. Intensification of grassland and forage use: driving forces and constraints. *Crop Pasture Sci.* **65**, 524 (2014).
134. Paudel, S. et al. Intensification differentially affects the delivery of multiple ecosystem services in subtropical and temperate grasslands. *Agric. Ecosyst. Environ.* **348**, 108398 (2023).
135. Byerlee, D., Stevenson, J. & Villoria, N. Does intensification slow crop land expansion or encourage deforestation? *Glob. Food Secur.* **3**, 92–98 (2014).
136. Pellegrini, P. & Fernández, R. J. Crop intensification, land use, and on-farm energy-use efficiency during the worldwide spread of the green revolution. *Proc. Natl Acad. Sci. USA* **115**, 201717072 (2018).
137. Lambin, E. F. & Meyfroidt, P. Global land use change, economic globalization, and the looming land scarcity. *Proc. Natl Acad. Sci. USA* **108**, 3465–3472 (2011).
138. Lam, S. K. et al. Mitigating soil greenhouse-gas emissions from land-use change in tropical peatlands. *Front. Ecol. Environ*. https://doi.org/10.1002/fee.2497 (2022).
139. Saha, D., Basso, B. & Robertson, G. P. Machine learning improves predictions of agricultural nitrous oxide (N_2O) emissions from intensively managed cropping systems. *Environ. Res. Lett.* **16**, 024004 (2021).
140. Akbarzadeh, Z., Maavara, T., Slowinski, S. & Van Cappellen, P. Effects of damming on river nitrogen fluxes: a global analysis. *Glob. Biogeochem. Cycles* https://doi.org/10.1029/2019GB006222 (2019).
141. Davidson, E. A. & Winiwarter, W. Urgent abatement of industrial sources of nitrous oxide. *Nat. Clim. Change* **13**, 599–601 (2023).
142. Xu, C. et al. Future material demand for automotive lithium-based batteries. *Commun. Mater*. **1** (2020).
143. Spears, B. M., Brownlie, W. J., Cordell, D., Hermann, L. & Mogollón, J. M. Concerns about global phosphorus demand for lithium-iron-phosphate batteries in the light electric vehicle sector. *Commun. Mater.* **3**, 14 (2022).
144. Sánchez, P. A. Tripling crop yields in tropical Africa. *Nat. Geosci.* **3**, 299–300 (2010).
145. Zou, T., Zhang, X. & Davidson, E. A. Global trends of cropland phosphorus use and sustainability challenges. *Nature* **611**, 81–87 (2022).
146. Hirel, B., Tétu, T., Lea, P. J. & Dubois, F. Improving nitrogen use efficiency in crops for sustainable agriculture. *Sustainability* **3**, 1452–1485 (2011).
147. Gollehon, N., Kellog, R. & Moffitt, D. Estimates of recoverable and non-recoverable manure nutrients based on the census of agriculture — 2012 results (USDA, 2016).
148. Dinda, S. Environmental Kuznets curve hypothesis: a survey. *Ecol. Econ.* **49**, 431–455 (2004).
149. Nepstad, D. et al. Frontier governance in Amazonia. *Science* **295**, 629–631 (2002).
150. Northrup, D. L., Basso, B., Wang, M. Q., Morgan, C. L. S. & Benfey, P. N. Novel technologies for emission reduction complement conservation agriculture to achieve negative emissions from row-crop production. *Proc. Natl Acad. Sci. USA* **118**, e2022666118 (2021).
151. Springmann, M. et al. Options for keeping the food system within environmental limits. *Nature* **562**, 519–525 (2018).
152. Kanter, D. R. et al. Nitrogen pollution policy beyond the farm. *Nat. Food* **1**, 27–32 (2020).
153. Zhang, X. et al. Quantification of global and national nitrogen budgets for crop production. *Nat. Food* **2**, 529–540 (2021).
154. Lim, T. et al. Increasing the value of animal manure for farmers (USDA, 2023).
155. Dou, Z., Toth, J. D. & Westendorf, M. L. Food waste for livestock feeding: feasibility, safety, and sustainability implications. *Glob. Food Secur.* **17**, 154–161 (2018).
156. Bergtold, J. S., Ramsey, S., Maddy, L. & Williams, J. R. A review of economic considerations for cover crops as a conservation practice. *Renew. Agr. Food Syst.* **34**, 1–15 (2019).
157. Basso, B., Shuai, G., Zhang, J. & Robertson, G. P. Yield stability analysis reveals sources of large-scale nitrogen loss from the US Midwest. *Sci. Rep.* https://doi.org/10.1038/s41598-019-42271-1 (2019).
158. Cammarano, D. et al. Modeling spatial and temporal optimal N fertilizer rates to reduce nitrate leaching while improving grain yield and quality in malting barley. *Comput. Electron. Agric.* **182**, 105997 (2021).
159. Brilli, L. et al. Review and analysis of strengths and weaknesses of agro-ecosystem models for simulating C and N fluxes. *Sci. Total Environ.* **598**, 445–470 (2017).
160. Del Grosso, S. J. et al. Global scale DAYCENT model analysis of greenhouse gas emissions and mitigation strategies for cropped soils. *Glob. Planet. Change* **67**, 44–50 (2009).
161. Gu, B., Ju, X., Chang, J., Ge, Y. & Vitousek, P. M. Integrated reactive nitrogen budgets and future trends in China. *Proc. Natl Acad. Sci. USA* **112**, 8792–8797 (2015).
162. Lassaletta, L. et al. Nitrogen use in the global food system: past trends and future trajectories of agronomic performance, pollution, trade, and dietary demand. *Environ. Res. Lett.* **11**, 095007 (2016).
163. Bodirsky, B. L. et al. Reactive nitrogen requirements to feed the world in 2050 and potential to mitigate nitrogen pollution. *Nat. Commun.* **5**, 3858 (2014).
164. Doelman, J. C. et al. Quantifying synergies and trade-offs in the global water-land-food-climate nexus using a multi-model scenario approach. *Environ. Res. Lett.* **17**, 045004 (2022).
165. Kyle, P. et al. Assessing multi-dimensional impacts of achieving sustainability goals by projecting the sustainable agriculture matrix into the future. *Earths Future* https://doi.org/10.1029/2022EF003323 (2023).
166. Li, T. et al. A hierarchical framework for unpacking the nitrogen challenge. *Earths Future* https://doi.org/10.1029/2022EF002870 (2022).
167. Ma, L. et al. Modeling nutrient flows in the food chain of China. *J. Environ. Qual.* **39**, 1279–1289 (2010).
168. Nazari Sharabian, M., Ahmad, S. & Karakouzian, M. Climate change and eutrophication: a short review. *Eng. Technol. Appl. Sci. Res.* **8**, 3668–3672 (2018).
169. Meerhoff, M. et al. Feedback between climate change and eutrophication: revisiting the allied attack concept and how to strike back. *Inland Waters* https://doi.org/10.1080/20442041.2022.2029317 (2022).
170. Popp, J. et al. Biofuels and their co-products as livestock feed: global economic and environmental implications. *Molecules* **21**, 285 (2016).
171. Baker, J., Murray, B., McCarl, B., Rose, S. & Schneck, J. Greenhouse gas emissions and nitrogen use in U.S. agriculture: historic trends, future projections, and biofuel policy impacts (Duke University Nicholas Institute for Environmental Policy Solutions, 2011).
172. Meinshausen, M. et al. Historical greenhouse gas concentrations for climate modelling (CMIP6). *Geosci. Model. Dev*. https://doi.org/10.5194/gmd-10-2057-2017 (2017).
173. Hoesly, R. M. et al. Historical (1750-2014) anthropogenic emissions of reactive gases and aerosols from the Community Emissions Data System (CEDS). *Geosci. Model. Dev*. **11**, 369–408 (2018).
174. Gütschow, J. et al. The PRIMAP-hist national historical emissions time series. *Earth Syst. Sci. Data* https://doi.org/10.5194/essd-2016-12 (2016).
175. Crippa, M. et al. GHG emissions of all world countries (Publications Office of the European Union, 2023).
176. Zhang, X. et al. Decarbonization in a eutrophic world. *Zenodo* https://zenodo.org/doi/10.5281/zenodo.11097253 (2024).

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Author contributions

X.Z. conceptualized the article. All authors contributed to the review of five decarbonization strategies (DS), with L.R. leading DS1, R.S. and W.Y. leading DS2, P.K. and H.N. leading DS3, E.A.D. leading DS4, and X.Z. leading DS5. X.Z., J.S.B. and W.Y. led the data synthesis and visualization. H.N. and E.A.D. led the development of Fig. 2 and its caption. All authors reviewed, revised and approved the final version of the draft.

Competing interests

The authors declare no competing interests.

Additional information

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