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Food Security Versus Environmental Health: The Nitrogen Dilemma

Henrike Mielenz[1] | Norbert Bischoff[2] | Klaus Butterbach-Bahl[3,4] | Uwe Häußermann[5] | Kurt-Christian Kersebaum[6,7,8] | Karin Rather[9] | Burkhard Stever-Schoo[1] | Reinhard Well[10] | Claas Nendel[6,11,8]

[1]Julius Kühn Institute—Federal Research Centre for Cultivated Plants, Institute for Crop and Soil Science, Braunschweig, Germany | [2]State Authority for Mining, Energy and Geology, Hanover, Germany | [3]Institute of Meteorology and Climate Research, Department Atmospheric Environmental Research (IMK-IFU), Karlsruhe Institute of Technology, Garmisch-Partenkirchen, Germany | [4]Pioneer Center Land-CRAFT, Department of Agroecology, University of Aarhus, Aarhus, Denmark | [5]Department of Landscape Ecology and Resources Management, Justus Liebig University, Giessen, Germany | [6]Leibniz Centre for Agricultural Landscape Research, Müncheberg, Germany | [7]Department of Crop Sciences, Georg August University, Göttingen, Germany | [8]Global Change Research Institute CAS, Brno, Czech Republic | [9]State Horticultural College and Research Institute Heidelberg, Heidelberg, Germany | [10]Thünen Institute of Climate-Smart Agriculture, Braunschweig, Germany | [11]Institute of Biochemistry and Biology, University of Potsdam, Potsdam, Germany

Correspondence: Henrike Mielenz (Henrike.Mielenz@julius-kuehn.de)

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ABSTRACT

Nitrogen is an essential element for plant growth and, consequently, crop production. However, the cycling of nitrogen in the environment can have negative effects. Here, we review the historical development of scientific and public awareness of these effects and the influence this has had on political action to control them. Focusing on Germany as an example, we explore past nitrogen myths and how scientific progress has dispelled them, contextualizing our current position as a society seeking sufficient food, a clean environment, and an end to climate change. Given the continuous heavy burden placed on farmers to solve all these issues, we argue that from the state that we have reached today, further reducing nitrogen inputs for crop production may not necessarily lead to a further reduction in nitrogen cycling in the environment but rather to reductions in yields and product quality. Nevertheless, there is still potential to improve the efficiency of nitrogen use for stabilizing crop yields and to increase resilience against potential harm to the environment and human health. We make a couple of suggestions about what can be done.

1 | Introduction

Nitrogen is the most critical essential nutritional element for plants that cannot be retrieved directly from the atmosphere (Gojon 2017). Provisioning of nitrogen to crop plants has boosted crop yields even before humans were aware of its existence (Bogaard et al. 2013). Animal excreta have played a key role in keeping soils fertile for millennia, and their recycling in fields for the benefit of plant production has been an important pillar in farming concepts worldwide. The identification of nitrogen and the technical knowledge of synthesizing ammonium and nitrate in mineral form and urea marked the inception of industrial agriculture (Erisman et al. 2008; Lotka 1925). The increased availability of mineral fertilizers enabled a significant increase in crop yields. This, in turn, triggered a sharp rise in population in the respective countries and, later, worldwide. However, the immense intensification of agricultural productivity has been accompanied by unwanted side effects. The production process for mineral fertilizer is energy-demanding and currently contributes 0.4–0.6 Gt CO_2eq y^{-1} to the total CO_2 emissions from fossil fuels (Gao and Cabrera Serrenho 2023; Tubiello et al. 2022). At the same time, agroecosystems soon became overloaded with nitrogen, and nitrogen losses into water bodies, the atmosphere, and adjacent ecosystems became a widespread problem for the environment and human health.

While the assumption that nitrate in food and drinking water causes cancer in the human digestive tract and cyanosis in infants stirred scientists' attention in the first half of the 20th century, the environmental aspect only became a topic of debate in the 1980s. High nitrogen concentrations were identified as problematic for aquatic ecosystems, especially in conjunction with phosphoric loads entering water bodies, causing heavy eutrophication and the collapse of many lake ecosystems (Vollenweider 1968). In addition, gaseous emissions in the form of ammonia (NH_3) and subsequent deposition were found to play a role in the acid rain period, when predominantly strong sulfur emissions through industrial processes, as well as ammonia emissions from agriculture, led to high concentrations of sulfuric and nitric acid in rainwater that caused a massive forest dieback in the 1980s. Not much later, in the upwelling climate change debate, the role of nitrogen in agroecosystems as a source of gaseous emissions, including nitrous oxide (N_2O) as a critical greenhouse gas (GHG) and stratospheric ozone-destroying substance (Crutzen 1970; Ravishankara et al. 2009), became increasingly prominent. N_2O emissions are currently estimated to be 18 Tg N per year globally, originating predominantly from agriculture (Tian et al. 2024). Although N_2O emissions are much smaller than CO_2 emissions, their reactive pattern in the atmosphere multiplies their warming potential by a factor of 273 (IPCC 2021). They have contributed approximately 6% to overall anthropogenic global warming since the preindustrial era (Tian et al. 2024). At that time, NH_3 emissions were identified as an important precursor for the formation of particulate matter in the air, which can be inhaled by humans, with potentially harmful consequences for the lungs (de Vries et al. 2024).

The intensive communication about the potential risks to human health posed by nitrates helped to make the wider public aware of the nitrate problem in the environment and motivated politicians to act against the large surplus of nitrogen in various ecosystems. This process was supported by scientific research about the different environmental fluxes and biochemical turnover processes and the environmental issues identified later on, in which other forms of nitrogen played a role. Soil scientists looked at this in conjunction with water movement in soils, sorption/desorption processes, and the turnover of organic matter. Agricultural scientists revisited existing recommendations for nitrogen fertilizer dosages and implemented ideas on how to account for the nitrogen already available in soils when applying fertilizer as a strategy to minimize nitrogen losses to the environment. Simulation models were developed to help test different hypotheses on the interplay of the different processes involved.

Today, it seems that we have reached the point where most of the relevant environmental processes that lead from nitrogen additions in soils to aqueous and gaseous emissions to the environment are understood, and the feedback mechanisms have been elucidated. However, the quantification of nitrogen losses remains challenging, and success stories on reducing losses seem scarce. From the global perspective, nitrogen pollution and the threat to pristine ecosystems and their biodiversity are considered one of the most pressing societal challenges. In the concept of planetary boundaries (Rockström et al. 2009a, 2009b), the carrying capacity of the planet's ecosystems for nitrogen has already been exceeded (Steffen et al. 2015). However, it is obvious that the different regions and governments of the world are at different stages in the fight against excess nitrogen (Kahiluoto et al. 2024; Schulte-Uebbing et al. 2022; Zhang et al. 2015), with a large number of obstacles to overcome. Germany is among the countries with the longest history of the nitrogen debate, and the long trajectory of implementing science-informed measures has led to a much lower nitrogen surplus today. Nevertheless, the European Commission's goal of reducing nitrogen pollution from agriculture to a level that is no longer harmful to health and the environment (EC 2021) seems unattainable for the agricultural sector, as any further reduction in nitrogen emissions would apparently require a reduction in nitrogen use, which would often have yield-reducing consequences for plant production. However, the problem is much more complex, and revisiting Germany's historical efforts in this regard probably provides a good example of what has worked thus far and what has not.

In this article, we provide an overview of the historical development of the use of nitrogen in agriculture, the environmental and health issues emerging from it, the political debate and administrative action in Germany, and the scientific work that has accompanied these processes (Figure 1). We highlight the most important overarching influences and scientific innovations that have led us to where we are today: do we have to accept reductions in crop yields to further reduce nitrogen losses, or is there still scope for becoming even more nitrogen-efficient in German agriculture?

2 | Nitrogen as a Driver of Crop Production

Before the invention of the Haber–Bosch process for industrial nitrogen fertilizer production, the application of mineral fertilizers to agricultural land was low and mainly originated from Chilean nitrate (60%) and ammonium sulfate from coking ovens (26%) (Addiscott 2005). Calculations by Smil (2001) determined that the world production of mineral N fertilizer rose from 80 kt N year^{-1} in 1860 to 360 kt year^{-1} in 1900. Patented in 1911 (Haber and Bosch 1911), the contribution of the Haber–Bosch process rose within 50 years to more than 85% of the world's total mineral nitrogen fertilizer production (Addiscott 2005; Smil 2001), with an exponential growth of total N production until the 1980s (Figure 2A).

The increasing use of nitrogen in agriculture has led to an increase in crop production and N uptake globally (Figure 3A; Ludemann et al. 2024). However, extensive fertilizer use in industrial countries has led to an increase in the nitrogen surplus (Figure 3B,C) accompanied by reduced nitrogen use efficiency (Zhang et al. 2015).

Industrial nitrogen production dropped in Europe in the late 1980s, driven by the demise of the Soviet Empire and the associated major political and socioeconomic upheavals and by environmental regulations (Figure 2A). In North America, nitrogen production decreased in the mid-1990s, while increasing production continued in Asia, with a major contribution from

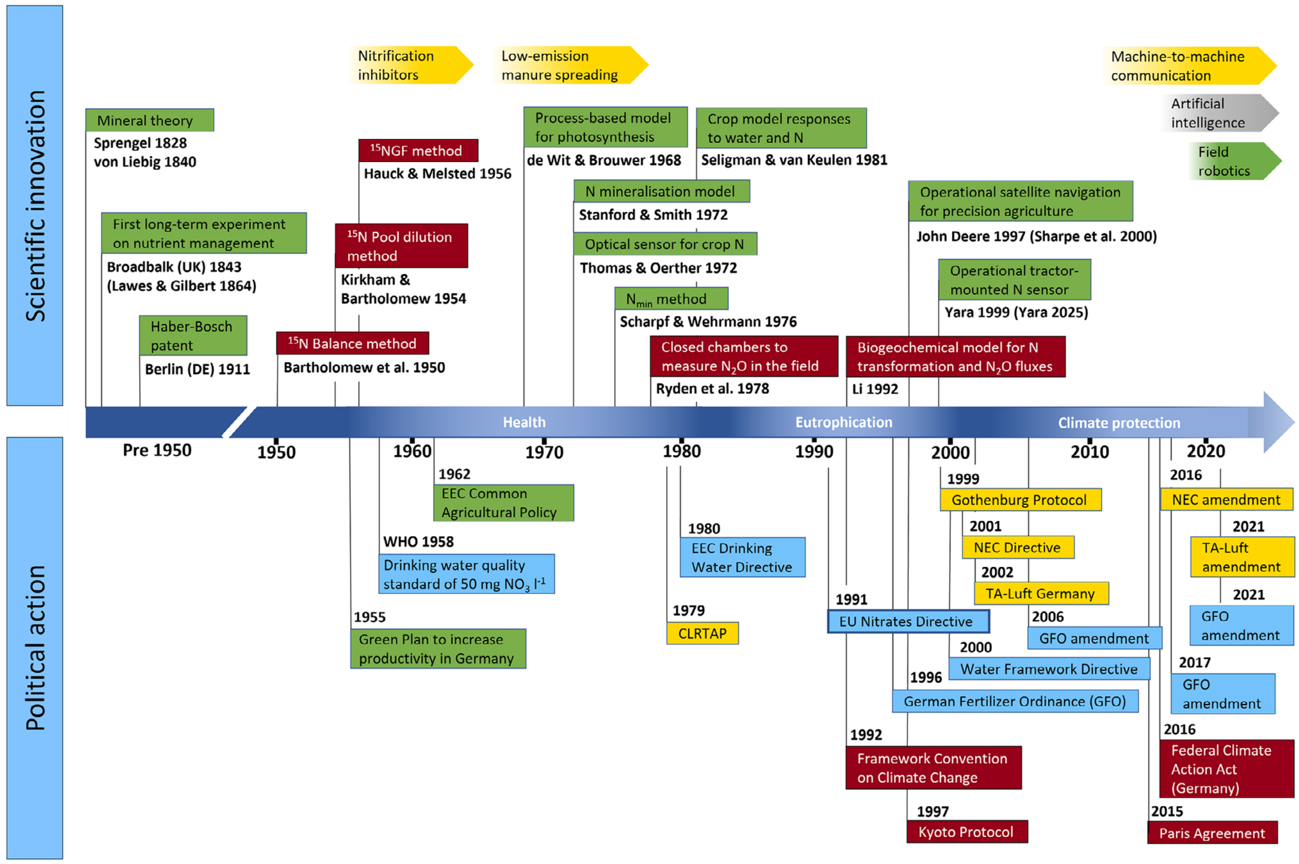

FIGURE 1 | Major milestones marking scientific innovation and political action in the history of society dealing with nitrogen. Colors refer to the environmental realm to which nitrogen is associated: blue = water quality, yellow = air quality, dark red = climate protection, green = crop production.

China until 2015 (Figure 2A). Since then, the Chinese government has paid more attention to solutions for environmental problems (Shen et al. 2024).

As a consequence of the Second World War, Europe suffered from massively destroyed infrastructure, including agriculture. While labor was scarce and seeds and machinery were lacking, an exceptionally cold winter in 1946–1947 followed by a heavy drought during summer 1947 with record-low crop yields (Figure 4B) induced a dramatic food shortage, with several hundred thousand famine deaths across large parts of central Europe (Gimbel 1968; Stanley 2018). The rationing of food continued into the 1950s, and food security in Europe was a high priority. In 1955, the West German government adopted the "Green Plan," a package of measures to promote the revival of agriculture, which included guaranteed income for farmers for grain, milk, and meat (Land- und Hauswirtschaftlicher Auswertungs- und Informationsdienst 1956). The following year, the Treaty of Rome was signed, committing the six founder states of the European Community (West Germany, France, Italy, the Netherlands, Belgium, and Luxembourg) to the Common Agricultural Policy (CAP). With the aims of increasing productivity, stabilizing markets, ensuring regular food supply, and guaranteeing farmers a fair standard of living while prices for consumers are reasonable, the CAP introduced "intervention buying." This gave farmers the incentive to produce, irrespective of the existence of a market for their produce (Addiscott 2005), and resulted in overproduction, for example, "butter mountains" and "wine lakes." The main direct effect was that guaranteed prices made it economically viable to use more nitrogen fertilizer for crop production (Figure 3D), which also enabled farmers to increase animal stocks (Figure 2B).

In Germany, the introduction of mineral N fertilizers allowed the use of an increasing number of marginal sites for agriculture. In addition to the increase in nitrogen fertilization, the economic incentives of CAP also led to a continuous conversion of grassland into arable land through plowing, often in conjunction with "necessary soil improvement" such as drainage. High nitrogen mineralization rates in the first years after conversion contributed to nitrate pollution of water bodies. The share of arable land within the agriculturally used land increased from approximately 58% to 71% in the period from 1950 to 2014 (Gömann et al. 2015). While arable land was reduced mainly in expanding conurbations, a noticeable expansion of arable land took place mostly at the expense of grassland, especially in northwestern Germany (Figure 4A for Lower Saxony) and parts of Bavaria (Gömann et al. 2015).

In Lower Saxony (and its former provinces), agriculturally used land (arable land and pasture) decreased continuously from 1878 to the present (Figure 4A). This federal state was selected as a representative for all of Germany because it had no significant changes in its enveloping borders over the two world wars. The drastic reduction in grassland in the 1880s (Figure 4A) could be an effect of the start of peat extraction for use as fuel (Blankenburg 2015). During the first half of the 20th century,

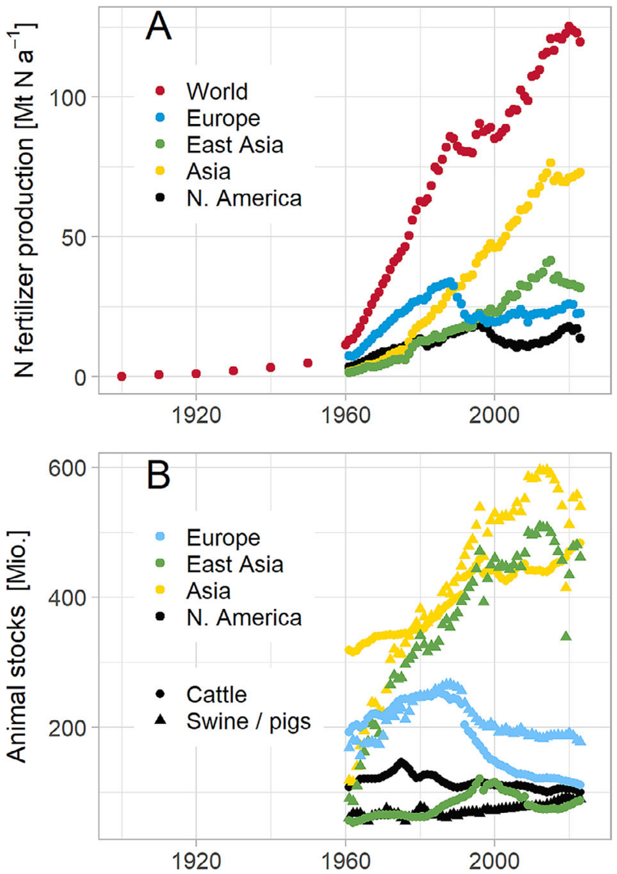

FIGURE 2 | (A) Nitrogen (N) fertilizer production 1900–2023 and (B) cattle and swine stocks 1961–2023 in different regions (data: 1900–1960: Smil (2011); 1961–2023: https://www.fao.org/faostat/en/#data, accessed November 7, 2025).

the agricultural land as well as the relation between arable land and grassland remained nearly constant at 54%–59% in Lower Saxony. However, the acreage of arable land increased by approximately 27% (approximately 393 000 ha) between 1950 and 2023, while permanent grassland was reduced by approximately 44% (approximately −547 000 ha), changing the percentage of arable land from 54% to 73% (Figure 4A). At the same time, there was a change in the share of the main arable crops (wheat, barley, oats, potatoes, maize, winter rape) from 1930 to 2023 in Lower Saxony. Oats (as feed for horses) had a share of 20% on arable land, but wheat only accounted for 8% in 1930 (2023: oats 0.6%, wheat 21%), while maize appeared with small acreage in the late 1930ies and increased to 30% in 2023 (see references in Figure 4).

Figure 4B shows the increase in wheat yields for Lower Saxony during the same period. While until the end of the 1950s, the increasing trend followed the long-term genetic trend estimated by Ahrends et al. (2018), the slope increased more strongly than the genetic trends estimated by Ahlemeyer and Friedt (2011) and Laidig et al. (2017) from 1960 until the 1990s (Figure 4B), which is attributed to the increasing use of mineral nitrogen fertilizers. Although the genetic trend remains positive, a decline in yield increase and yield leveling occurred in Lower Saxony as well as in Germany and many countries in the world from the 1990s on (Grassini et al. 2013). Several authors discussed the climate change complex but also mentioned political and economic incentives, which might lead to an extensification of agriculture and therefore be responsible for yield stagnation (Brisson et al. 2010; Lin and Huybers 2012).

While the increasing use of mineral fertilizers led to a substantial increase in crop yields, nutrient balances showed increasing nitrogen surpluses in cropland (Figure 3). In early nutrient balance calculations for agricultural land (cropland and pasture) at the national scale, the nitrogen surplus of nitrogen input from fertilizers and N deposition over the nitrogen output through crop harvest products amounted to more than 100 kg N ha^{-1} $year^{-1}$ in the beginning of the 1980s for the western part of Germany (Bach 1985; Bach and Frede 1998). The nitrogen surplus was similar in the former German Democratic Republic (Nieder et al. 2007). In the 1990s, fertilizer use and surplus declined in Europe and Germany (Figure 4B,C). This was due to the collapse of structures in the former Warsaw Pact countries, including East Germany, which also led to a significant decline in livestock numbers (Figure 2B) and to environmental regulations (Council of the EU 1991).

3 | Rising Concerns About Nitrogen as a Threat to Human Health

Nitrate has always been considered a chemical compound non-toxic to human health. Therefore, the addition of increasing amounts of nitrates to soils has never been questioned, especially not against its promotional effect on crop yields. However, in 1902, "blue baby syndrome," an oxygen deficiency of infants (methemoglobinemia; Stokvis 1902; Talma 1902), was first described, and nitrite, a nitrate metabolite, was hypothesized to be responsible for the death of a number of bottle-fed infants (Comly 1945). This drastically changed the perspective on nitrate, and its intake by humans via water and food became a serious issue from the 1950s onward.

3.1 | Drinking Water as a Source of Human Nitrate Intake

Nitrite ions can inactivate hemoglobin in the blood, thereby lowering its oxygen-carrying capacity. This process results in affected people developing blue skin. Infants under three months, whose stomach pH value is still variable, have a higher chance of nitrite formation in their gastrointestinal tract from nitrate, which they may have consumed with infant formula. Drinking water as a possible source of nitrate for methemoglobinemia was identified in 1940 (Schwartz and Rector 1940), and after more cases were reported (Campbell 1960; Downs 1950), the World Health Organization took action and set a 50 mg NO_3 L^{-1} threshold value in its first recommendation for quality standards for drinking water (WHO 1958).

The carcinogenic potential of nitrate is based on a possible conversion to nitrite by salivary bacteria and a subsequent reaction to N-nitroso compounds in the acidic environment of the stomach (Dieter et al. 2011; Nieder and Benbi 2022). N-Nitroso compounds and cancer formation dominated medical research in the mid-1960s and were focused on nitrosamines (Jobst 1990). In contrast to nitrite in cheese and meat products, science did not

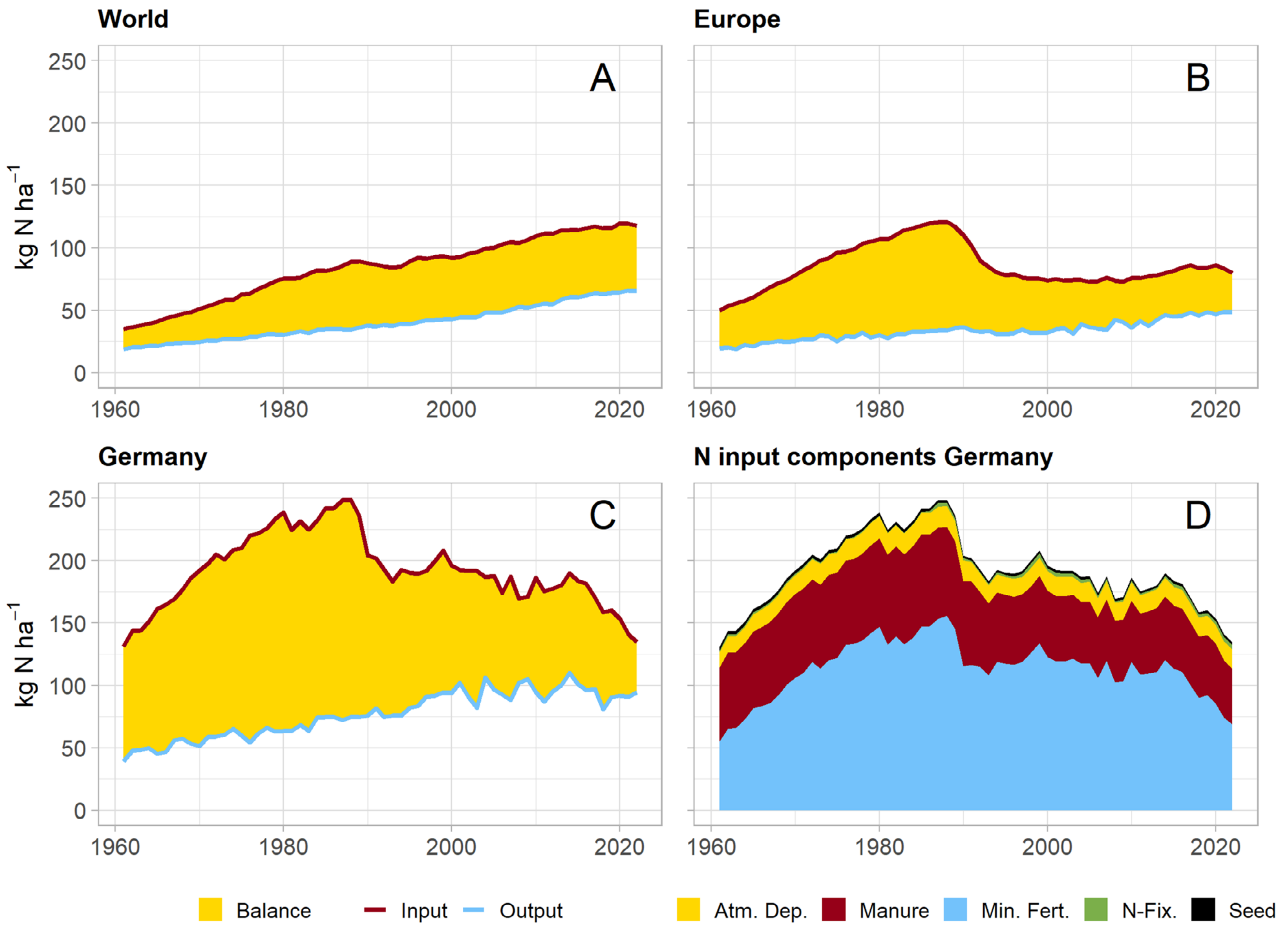

FIGURE 3 | Nitrogen input (mineral and organic fertilizers, N deposition, seeds, and biological N fixation), output (crop removal) and the resulting N field balance for cropland in (A) the world, (B) Europe, (C) Germany, and (D) the input components for Germany 1961–2022 (data: https://www.fao.org/faostat/en/#data/ESB, accessed November 25, 2025).

accept the important role of nitrate until the mid-1970s, when cancer formation gradually replaced the methemoglobinemia discussion. This was supported by recorded methemoglobinemia cases associated with high-nitrate drinking water from private wells, which also often had high bacterial levels (Comly 1945; Jobst 1990).

Further research in the 1980s uncovered the nitrate–nitrite–nitric oxide pathway and revealed that nitrate is produced endogenously in the body (Lundberg 2012). Consequently, during the 1990s, the perspective shifted from viewing nitrates solely as harmful to recognizing their potential therapeutic benefits (Bowles et al. 2025). The 2000s saw new studies emerging where long-term exposure to nitrate in drinking water was found to be significantly associated with an increased risk of cancer, even at nitrate concentrations well below the drinking water standard of 50 mg L^{-1} (e.g., Jacobsen et al. 2024; Picetti et al. 2022; Schullehner et al. 2018; Stayner et al. 2021; Ward et al. 2018). Consequently, an international expert group appointed by the Danish government recently recommended substantially lowering the nitrate limit in drinking water to 6 mg L^{-1} (Timmermann et al. 2026). The Danish government intends to follow this recommendation (Heissel 2026).

3.2 | Vegetables as Source of Human Nitrate Intake

Spinach moved into focus when scientists found that in previously cooked spinach, bacteria would gradually transform nitrate into nitrite when the spinach was rewarmed after storage (Sinios and Wodsak 1965). During growth, spinach accumulates concentrations of nitrates well beyond its physiological demand, if supplied, which potentially leads to considerable nitrite levels.

The European Food Safety Authority (EFSA) has identified vegetables and fruits as the primary sources of dietary nitrate intake, contributing 50%–75% of the overall intake (EFSA 2008). An acceptable daily intake (ADI) for nitrate of 3.7 mg kg body weight^{-1} day^{-1} was established by the former Scientific Committee on Food (SCF) and was reconfirmed by the Joint FAO/WHO Expert Committee on Food Additives (JECFA) in 2002. The European Union set maximum levels for nitrates in certain vegetables (European Commission 2011; Santamaria 2006). However, in contrast to drinking water, nitrate-rich vegetables do not show any positive correlation with cancer risk (Bondonno et al. 2023; Nöthlings et al. 2008; Tan et al. 2024). Vegetables are rich in antioxidants, which may inhibit

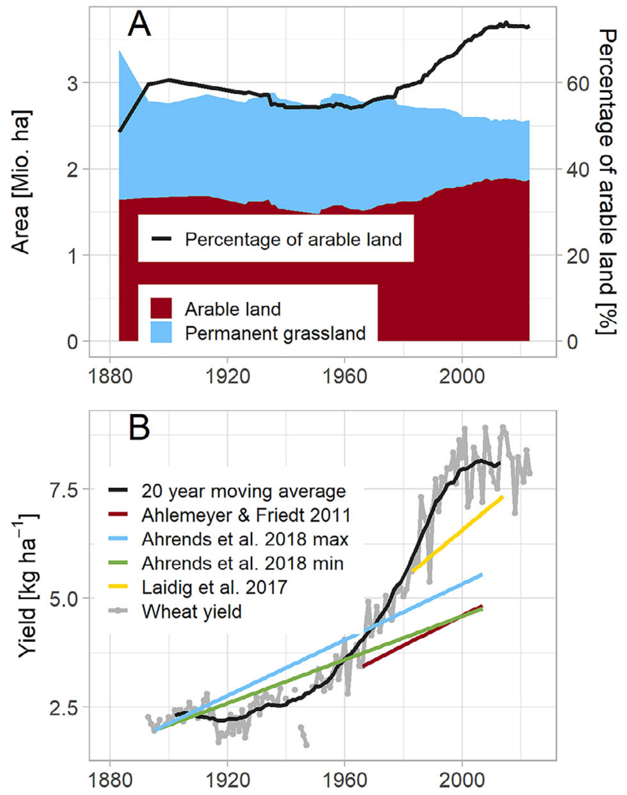

FIGURE 4 | Development of (A) area of arable land and permanent grassland and the contribution of arable land to agricultural used land from 1878 to 2023 and (B) wheat yields from 1893 to 2023 in Lower Saxony, Germany (*source*: Kaiserliches Statistisches Amt 1883–1918, LSN 2006–2023, Statistik der Britischen Besatzungszone 1945–1949, Statistisches Bundesamt 1953–2005, Statistisches Reichsamt 1919–1943). Lines show genetic trends derived either from variety trials or comparisons of varieties with different appearance years (Ahlemeyer and Friedt 2011, Ahrends et al. 2018, Laidig et al. 2017).

the formation of harmful N-nitroso compounds. EFSA therefore concluded that the benefits of consuming vegetables outweigh the potential risks associated with nitrate exposure, and their consumption is encouraged as part of a healthy diet (EFSA 2008). However, various authors have highlighted the need for more comprehensive and long-term studies to better understand the health impacts of dietary nitrate (Shannon et al. 2021).

4 | From the Human to the Ecosystem Perspective

Protecting human health has been the most important driver for designing and implementing policies and measures that are aimed at reducing nitrogen loads in drinking water and food during most of the 20th century. However, during the 1980s, the perspective shifted gradually. While in medicine, the benefits of nitrates to the human body were identified and weighed against the very rare cases of methemoglobinemia and the contradictory relations between nitrate intake and the development of cancer, the role of nitrogen in a range of aggravating environmental problems became increasingly evident toward the end of the 20th century. Between 2010 and 2014, agriculture accounted for 88% of NO_3 leaching in Germany (Häußermann et al. 2021), 93% of NH_3 emissions (UBA 2025c), 12% of NO_x emissions (UBA 2025c), and 74% of N_2O emissions in 2023 (UBA 2025d).

4.1 | Nitrogen as an Important Air Pollutant

In the 1960s, the acidification of Scandinavian lakes caused by sulfur dioxide emissions and extensive air pollution was recognized as a problem, even at very large distances from the point of emission (Belle and Johnson 2024; Likens 2021). This called for cooperation at an international level and led the UN ECE states to adopt the Convention on Long-Range Transboundary Air Pollution in 1979 (Geneva Convention on Air Pollution). During the 1970s and 1980s, air pollution by acid sulfur and nitrogen components increased massively, supporting smog (smoke + fog) incidents across major European cities and triggering the acid rain phenomenon, which caused Central European forests to die back in enormous quantities (Kandler and Innes 1995; Ulrich et al. 1979). The Gothenburg Protocol of 1999 (Deutscher Bundestag 2004) addressed eutrophication of water bodies for the first time and thereby brought ammonia more into focus. The Gothenburg Protocol and its revision were adopted into EU law with the directive on national emission ceilings (NEC) Directive (2001/81/EG, Council of the EU 2001) and the new NEC Directive (2016/2284/EU, Council of the EU 2016). Germany committed to reducing annual NO_x emissions by 65% and annual NH_3 emissions by 29% by 2030 compared to 2005 levels (BMJV 2018).

Agriculture has been identified as the most significant emitter of ammonia in Germany, with 527 kt NH_3 in Germany in 2023, corresponding to 93% of total ammonia emissions (Figure 5; UBA 2025c). The primary source of ammonia emissions from agriculture is animal husbandry, where emissions occur from holdings, storage, and spreading of farm manure (Rösemann et al. 2025). High NH_3 emissions also occur when certain mineral fertilizers or fermentation residues are applied (Figure 5; Kemmann et al. 2025; Rösemann et al. 2025), of which urea is the most prominent (Hu et al. 2024). Emissions from urea can be substantially mitigated through the application of urease inhibitors (Cameron et al. 2013; Helfrich et al. 2024; Ni et al. 2014; Schoof et al. 2025; Schraml et al. 2018; WHO 1984). Nonetheless, concerns regarding the fate, degradability, and environmental implications of the added substances remain unresolved (Biewald et al. 2025).

Measures identified as effective in reducing NH_3 emissions during manure application include the rapid incorporation of manure immediately after surface application, direct injection into the soil, and, in the case of standing crops, application in strips close to the ground (Hu et al. 2024; Rösemann et al. 2025). These practices were incorporated into the German Fertilizer Ordinance (GFO) (BMEL 2017b, 2020) and the Technical Guidance on Air Quality Control (TA LUFT, BMU 2021) and became part of the suite of measures implemented in the National Clean Air Program to achieve the NEC reduction targets for NH_3.

The use of broadcasters has notably declined on cropland, with low-emission methods becoming the preferred alternative (DESTATIS 2021; Rösemann et al. 2025). More recently, the acidification of farm fertilizers in holdings, storage, or during application has been identified as another effective reduction

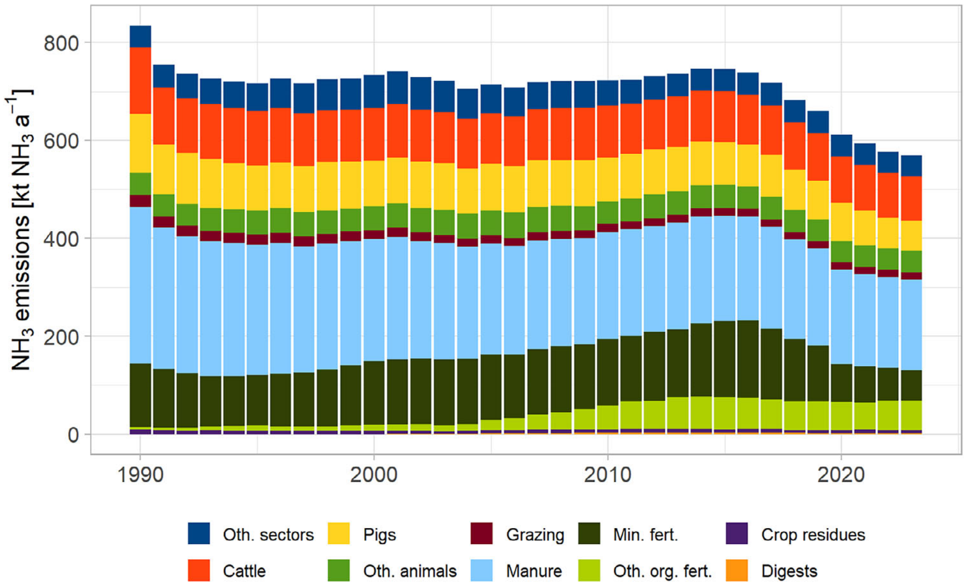

FIGURE 5 | Ammonia emissions in Germany from 1990 to 2023 (*source*: Rösemann et al. 2025). Emissions are subdivided into those originating from animal housing and manure storage (pigs, cattle, and other animals), grazing, application of manure, mineral and other organic fertilizers, crop residues, management of digests (storage), and sectors other than agriculture.

strategy (Fangueiro et al. 2015; Häußermann, Bach, et al. 2020; Huf et al. 2023). Slurry separation and subsequent application of the remaining liquid phase was also found to reduce NH_3 emissions, as it infiltrates much more quickly into the soil (Pedersen et al. 2024). Dilution of slurry with water at application has the same effect (Heinemann et al. 2024; Schmidhalter 2024) but might not be economically viable in practice.

Ammonia emissions remained relatively constant from 1991 to 2016. Since then, the implemented emission-reducing measures in combination with an overall decline in livestock numbers and reduced use of mineral fertilizers have led to falling ammonia emissions (Figure 5). Although the regulations in the GFO and TA Luft provide a significant contribution to achieving the NEC targets (Hu et al. 2024), they will not be sufficient on their own to achieve the 2030 targets (Häußermann, Klement, et al. 2020). In a recent study, NH_3 emissions after urea application were measured over three years across six agroecological regions in Germany. The findings revealed that the emission factors were significantly lower than the current IPCC (IPCC 2019) and EMEP (EEA 2023) emission factors (Kemmann et al. 2025). Implementing these measures in future inventories will lead to lower NH_3 emission levels.

4.2 | Nitrogen as a Pollutant in Aquatic Ecosystems

The quality of groundwater had already been a major source of drinking water, and lake acidification had already sensitized science to telecoupling pathways of ecosystems. However, the increasing influx of nutrients into surface waters gave rise to another dramatic environmental phenomenon: the collapse of lake ecosystems due to heavy eutrophication (Carpenter et al. 1998; Smith and Schindler 2009). Soon after, it became obvious that phosphorus and nitrogen were the main driving elements for this process. They caused a massive algae bloom in otherwise nutrient-limited waters, then a dieback and sinking of the biomass and a subsequent decomposition process that consumed all oxygen from the water, which was then unavailable for other organisms to live (Bhagowati and Ahamad 2019; Carpenter et al. 1998). While earlier EU legislation to protect drinking water resources was driven by the need to protect human health (Goodchild 1998), the issue of eutrophication motivated the design of the EU Nitrates Directive (91/676/EEC; Council of the EU 1991) and later the Water Framework Directive (WFD, 2000/60/EC, EP; Council of the EU 2000), which aimed to ensure that all European water bodies achieved at least good chemical and ecological status by 2027. In addition, the European Green Deal of 2020 contains an element of reducing nutrient losses by at least 50% by 2030, using 2012–2015 as the baseline.

The EU Nitrates Directive sets out a number of steps to be fulfilled by member states. This includes the monitoring of nitrate concentrations in ground and surface waters in nitrate-vulnerable zones (NVZs) and the establishment of nitrate action programs with mandatory measures to reduce and prevent water pollution, which in Germany is primarily implemented through the GFO. Targeted management regulations and advisory programs have been recommended for drinking water catchments by federal state authorities (Land Baden-Württemberg 2001) and private consulting (Kremer and Rathing 2021; Zander et al. 2015).

Historic measurements of aquifer nitrate concentrations are rare in Germany. Haakh (2018) showed an increase in NO_3 concentrations in the aquifer of the Donauried between 1930 and 2017 from approximately 10–15 to 30–40 mg NO_3 L^{-1} with a steeper slope during 1960–1995. The increasing nitrate concentrations in single drinking water wells raised awareness of nitrate pollution from diffuse sources such as agricultural land. This led to the establishment of a systematic monitoring network in the early 1980s, operated by the permanent working group on water issues

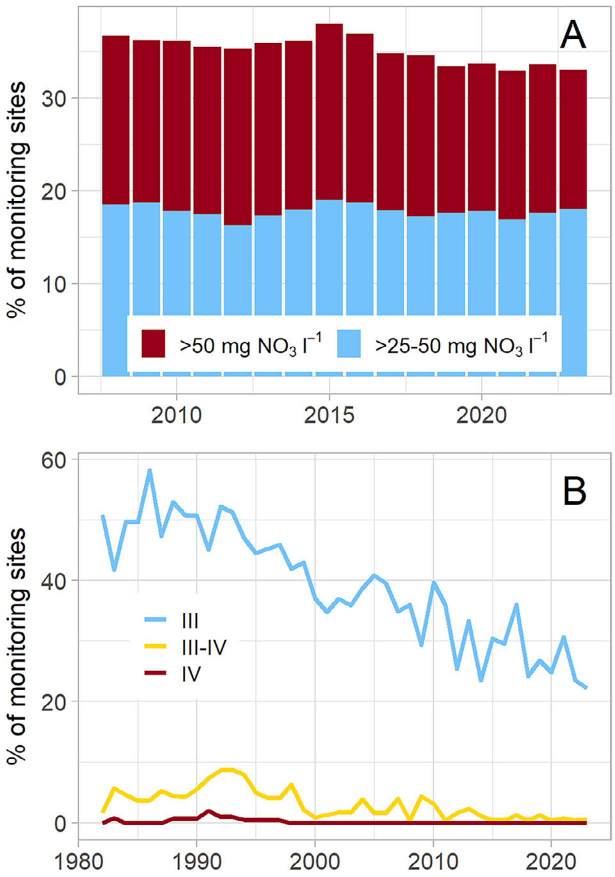

FIGURE 6 | (A) Percentage of wells of the European Environmental Agency (EEA) monitoring network in Germany for nitrate in groundwater with critical NO_3 concentrations 2008–2023 (*source*: UBA 2025a). (B) Percentage of the German monitoring network for nitrate in surface water with critical NO_3 concentrations, 1982–2023. III: 22.15–44.30, III–IV: 44.30–88.60, IV: > 88.60 mg NO_3 L^{-1} (44.3 mg NO_3 L^{-1} = 10 mg N L^{-1}) (*source*: UBA 2025b based on data from LAWA).

of the Federal States and the Federal Government (LAWA), represented by the Environmental Ministers' Conference.

In the mid-1990s, a monitoring network was established in Germany to document the effects of environmental measures on the upper levels of 181 aquifers under mainly agricultural land use. In the context of nitrate reporting under the Nitrates Directive, a larger network of 692 sites representative of agricultural inputs was created, of which 621 were consistently monitored (BMEL and BMUV 2024). This network is part of the European Environment Agency's (EEA) monitoring network, representing all types of land use. As sites have changed over time, obtaining a long-term overview of the nitrate situation in Germany is difficult. Over the last few decades, it has become clear that although a slight reduction in the number of EEA monitoring sites exceeding 50 mg NO_3 L^{-1} has been observed, one-third of the monitoring sites are still in critical condition, that is, > 25 mg L^{-1} (Figure 6A).

For surface waters, the German monitoring network increased from 120 to 140 locations during the 1980s and 200–230 in the 1990s to approximately 250 today. During the 1980s, more than 55% of the monitored sites showed at least an increased nitrate level and exceeded the target classes (Figure 6B). In contrast to groundwater, a significant reduction in average nitrate levels was observed in rivers at both the European and national levels over the period 1992–2012 (BMEL and BMUV 2024). However, since then, the improvement has slowed down (Figure 6B). While the water quality of surface waters can react within short periods to changes in N input to agricultural land, mainly due to nitrate from surface runoff, interflow, and NH_3 deposition, groundwater reacts with a significant delay due to long transport distances from the input source of nitrate to the groundwater observation points.

In the North Sea, marine waters consistently show nitrate concentrations below 0.49 mg NO_3 L^{-1}, while coastal waters range between 0 and 9.99 mg NO_3 L^{-1} (BMEL and BMUV 2024). In contrast, the Baltic Sea exhibits a different trend, with the majority of coastal waters and all marine waters within the German maritime borders falling within the 0–0.49 mg NO_3 L^{-1} nitrate concentration class. Despite recent improvements, coastal and marine waters remain in poor ecological conditions (BMEL and BMUV 2024).

In the amendment of the German Fertilization Act (DüngeG; BMEL 2017a), the added purposes of "sustainable and resource-efficient use of nutrients" and "avoiding nutrient losses to the environment as far as possible" lifted environmental concerns to a level equal to other purposes of the Fertilization Act for the first time. This and the adjustment of the GFO (BMEL 2017b) shifted fertilization strategies away from a purely economic to an ecological-economic optimum, with significantly lower fertilizer quantities to be used (Taube 2021). However, these efforts were deemed insufficient by the EU Commission. In 2018, the European Court of Justice found Germany guilty of violating the EU Nitrates Directive. In response, the federal government presented a new draft for the GFO in February 2020 that defined even stricter rules. A new component of the GFO 2020 (BMEL 2020) prescribed the designation of nitrate-polluted "red" areas and eutrophic "yellow" areas by the federal states if monitored water bodies exceed the quality standard of 50 mg NO_3 L^{-1} or are classified as eutrophic based on their phosphate levels, respectively. In these areas, stricter measures must be taken to reduce nitrogen and phosphorous inputs into water bodies. The EU Commission accepted this as sufficient and declared the treaty violation procedure closed in June 2023. However, a recent conviction against Germany resulting from a lawsuit filed by Environmental Action Germany (DUH) could lead to further amendments to German fertilizer legislation. The amendments to the GFO in 2017 and 2020 have already led to a significant reduction in nitrogen surplus (Figure 3D; BMLEH 2024). However, this reduction cannot yet be observed in ground or surface waters (Figure 6A, B), which may be caused by the dry conditions and low seepage rates of recent years in combination with the long transport distances from the location of nitrate input to the observation points in the groundwater. However, higher mineralization rates in the course of reduced N fertilization (N mining theory, e.g., Mahal et al. 2019) add to this phenomenon.

In the long term, the N surplus is the main driver for NO_3-leaching from agricultural fields to groundwater, and the N concentration of seepage water depends on dilution from the amount of seepage water (Fier and Thiermann 2024). Therefore, the N-surpluses have a greater impact in regions with low seepage rates than in regions with high seepage rates. Due to denitrification in soil, in the drainage zone, and in groundwater, not all of the N surplus ends up as NO_3 in wells, springs, and streams (Kunkel and Wendland 1997). In the rooting zone, energy sources for denitrification are replenished continuously. Therefore, denitrification here can be considered sustainable. In contrast, denitrification below the rooting zone is not sustainable because the energy sources for this process, that is, fossil stocks of organic carbon and sulfides, are gradually oxidized and thus depleted in the long term (Eschenbach et al. 2015; Fier and Thiermann 2024). Denitrifying conditions in groundwater are typically found in lowlands with Pleistocene and Holocene unconsolidated deposits, while the process is typically low or absent in solid rock aquifers due to the absence of suitable reductants (Wolters et al. 2022). In the drainage zone between the rooting zone and the groundwater table, NO_3- removal by denitrification is also possible when fossil electron donors are present in sediments (e.g., Funk et al. 1996; L. Li et al. 2023; Waterhouse et al. 2025). However, due to the current lack of information on the abundance of these conditions and because there are only very few direct measurements of denitrification in the vadose zone, the role of lowering NO_3- leaching into groundwater in Germany and worldwide still needs to be clarified.

4.3 | Nitrogen as an Important Driver of Global Warming

N_2O accounted for 4.8% of global GHG emissions in 2024, making N_2O the third most important anthropogenic GHG worldwide and from all sources (Crippa et al. 2025). Globally, fertilized croplands are a major source of the atmospheric GHG N_2O. Current estimates of direct N_2O emissions from agriculture are in the range of approximately 3.9 Mt N year^{-1} (2020) plus approximately 1.3 Mt N year^{-1} due to indirect emissions (Tian et al. 2024). Indirect emissions result from NH_3 emissions and subsequent N deposition to unmanaged land and from NO_3 leaching to aquatic ecosystems (Well and Butterbach-Bahl 2010). In 2024, the share of N_2O in GHG emissions in Germany was 3.5% overall but 34% in the agricultural sector; that is, 74% of N_2O emissions came from agriculture (UBA 2025d). Hereby, direct emissions from fertilizer application contribute approximately 30%, and indirect emissions contribute approximately 20% (Figure 7). From 1990 to 2023, N_2O emissions from agriculture in Germany decreased by 26% from 103.2 kt N_2O a^{-1} to 76.7 kt N_2O a^{-1} (Figure 7).

Atmospheric N_2O concentrations currently measure 340 ppb, which is approximately 60 ppb above preindustrial concentrations, and they rise at a rate of approximately 1% per year (WMO 2024). While the increase is well documented, source attribution remains difficult and is associated with large uncertainties (Palosuo et al. 2025; Tian et al. 2024). This is particularly true for emissions from agricultural soils, as emissions are not only affected by the rate of N fertilization and tend to increase exponentially when N fertilization exceeds crop N requirements (Shcherbak et al. 2014; van Groenigen et al. 2010) but also depend on other soil-specific environmental conditions, such as pH, texture, organic carbon content and mineralization activities (e.g., Butterbach-Bahl et al. 2013; Liu et al. 2014; Yao et al. 2024). The latter is particularly significant for drained organic soils (Figure 7; Säurich et al. 2019). Moreover, as soil N_2O emissions are mainly driven by the soil microbial processes of nitrification and denitrification (Butterbach-Bahl et al. 2013), their spatiotemporal pattern changes quickly (Butterbach-Bahl et al. 2022; Kravchenko et al. 2017; e.g., Mutegi et al. 2010; Schlüter et al. 2025), creating "hot moments" and "hotspots" (Groffman et al. 2009). These include freeze–thaw events (Luo et al. 2012; Wagner-Riddle et al. 2017) and rewetting of dry soils (Harrison-Kirk et al. 2013), with episodes of a few days potentially contributing 50%–80% of total annual emissions (Groffman et al. 2009). The challenge of accurately quantifying N_2O emissions from agricultural soils is still unresolved, as any further progress requires high-resolution, continuous, multiyear measurements of total N budgets and N_2O losses from representative agricultural sites (Palosuo et al. 2025). However, reliable methods for this purpose are currently not available.

The first international action to set a goal for stabilizing GHG concentrations to prevent dangerous anthropogenic interference with the climate system was the United Nations Framework Convention on Climate Change (UNFCCC; UN 1992) in 1992 (Um 2024). In 1997, the UNFCCC was supplemented by the Kyoto Protocol (UN 1998), with which binding GHG reduction targets were agreed upon for the first time. These were −5% of the 1990 emission levels for the industrialized countries between 2008 and 2012. The EU and Germany, however, committed to more ambitious targets of −8% and −21%, respectively (UBA 2024). With the 2015 Paris Agreement (UNFCCC 2015), it was agreed to limit anthropogenic global warming to below 2°C, preferably 1.5°C. The Paris Agreement led to further-reaching emission reduction targets with the goal of net zero emissions by 2050 at the EU level in the European Green Deal (EC 2019; EP, Council of the European Union 2021) and by 2045 in Germany in the Federal Climate Action Act (BMJV 2019).

N_2O fluxes typically increase with N surplus. Therefore, lowering N fertilization has been suggested as a general means to lower N_2O fluxes (van Groenigen et al. 2010). However, recent fieldwork has documented that it is most relevant in sites with high N_2O emission potential (Foldal et al. 2024). Further strategies to lower N_2O fluxes include the use of nitrification inhibitors, optimization of soil pH and organic fertilization, and specific management of intermediate crops and crop residues.

Nitrification inhibitors show significant potential for reducing direct N_2O emissions after fertilization at sites that tend to have high N_2O emissions (Flessa et al. 2014). However, their impact on annual N_2O emissions at different sites has not yet been sufficiently documented due to a lack of long-term studies (Hutchings et al. 2024).

The management of pH through liming is known to lower N_2O fluxes by supporting the N_2O reductase of denitrifiers (Wang et al. 2018). At the field scale, liming decreased N_2O emissions due to increased N_2O reduction via denitrification and by promoting plant growth (Abalos et al. 2020). A recent study showed that 41% of arable land and 52% of grassland soils have pH values lower than recommended (Müller et al. 2022). This indicates that more conscientious pH management of these soils could improve soil fertility and crop yield (Müller et al. 2022) and lower N_2O fluxes.

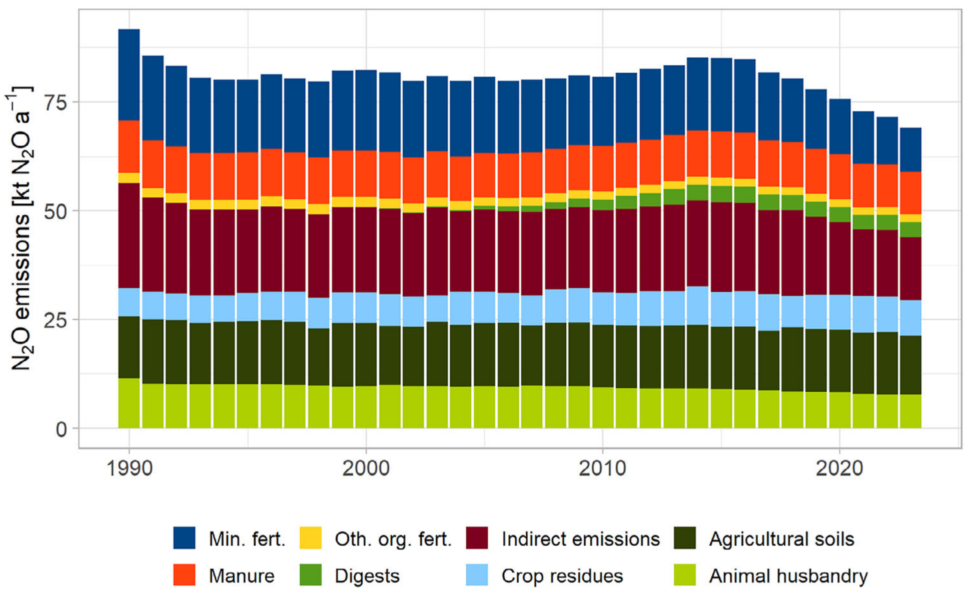

FIGURE 7 | Annual N_2O emissions from German agriculture from 1990 to 2023 (*source*: Rösemann et al. 2025). Emissions are subdivided into those originating from the application of mineral fertilizers, manure, and other organic fertilizers (including grazing), management of digests (storage, spreading), indirect emissions (due to leaching, runoff, deposition), crop residues, agricultural soils (degradation of managed organic soils, mineralization of soil organic matter), and animal husbandry (manure management).

Regarding organic fertilization, measures to reduce NH_3 emission, such as immediate incorporation through tillage or injection into the soil, can lead to higher N_2O fluxes from denitrification due to oxygen depletion during the respiration of labile C compounds (Webb et al. 2010). However, the effects of organic fertilization on N transformation are often inconsistent due to the complex interaction of soil and climate (Nyameasem et al. 2023).

Crop residue incorporation into the soil can similarly enhance denitrification and related N_2O fluxes (Abalos, Rittl, et al. 2022). Lowering of N_2O fluxes can be achieved by crop residue removal, shallow incorporation, incorporation of only residues with a high C:N ratio, and avoidance of the incorporation of immature residues (Abalos, Recous, et al. 2022). Winter catch crops have been considered a potential mitigation of N_2O fluxes by their ability to lower mineral N levels after harvest. However, recent research has demonstrated that the opposite is often the case (Helfrich et al. 2024; Nasser et al. 2025), apparently due to root respiration and rhizodeposition during catch crop growth as well as incorporation effects.

Progress in optimizing nitrous oxide (N_2O) mitigation through agricultural field management has been hindered by the difficulty of quantifying N_2O reduction to N_2. Thus, the intricate interplay of factors influencing the denitrification of nitrogen in arable soils remains largely unresolved. (Grosz et al. 2023). Although the role of microbial hotspots and their heterogeneous distribution in soil are generally accepted, their extent is still difficult to capture (Schlüter et al. 2025). Currently, field data to test the impact of N_2O mitigation options on N_2 fluxes are missing.

The annual agricultural emissions inventory is compiled to quantify the reduction in climate-relevant gases emitted from agricultural sources. (Rösemann et al. 2025). Herein, a range of approaches that integrate data from agricultural statistics, measurement data, emission factors, and modeling are applied (Mathivanan et al. 2021; Rösemann et al. 2025). Because the quantification of N_2O losses associated with agricultural land management has been based solely on approximate estimation methods and the spatial distribution of these measures is not explicitly known, emission inventories cannot sufficiently consider detailed management or environmental influences (Palosuo et al. 2025).

5 | Scientific Innovations and Practical Implications

Agronomy textbooks in the late 19th century have already promoted fertilization as a strategy to compensate for what the crop would consume and contained detailed calculations of how much nitrogen a crop harvest would remove from the system (e.g., Hotter 1898). Since then, the official fertilizer recommendations for crops (with the exception of grapevine, for which nitrogen uptake dynamics had hardly been studied until the 1980s) have not changed substantially with respect to yield expectations. However, the postwar decades in Europe—and in Germany specifically—created an economic environment that provided almost no barriers to nitrogen use to promote ever-increasing yields, and nitrogen was unconsciously applied by farmers. As public awareness of potential health problems and the eutrophication of lakes and rivers grew, the ideal of nitrogen-efficient crop production was revived.

The central idea of emerging concepts was to regard soil as a reservoir of nitrogen that could be utilized more wisely, and science started to investigate seasonal nitrogen dynamics more deliberately. This required determining the amount of dissolved mineral nitrogen (N_{min} resp. Nan) in the root zone at the start of the vegetation period by taking samples. The "N_{min} method" proposed the calculation of the first nitrogen fertilization rate in spring from the difference between the N_{min} content in the soil and a crop-specific target value (Scharpf and Wehrmann 1976; Wehrmann and Scharpf 1986). A similar approach was developed in parallel in the former German Democratic Republic (Müller et al. 1976). Here, the target values already included an average N release from the soil during the season. However, studies on fallow plots revealed significant variability in soil supply capacity (Köhler 1983), prompting the development of mathematical models to calculate the seasonal supply, for example, for field-specific fertilizer recommendations for vegetable crops (Fink and Scharpf 1993). The first GFO (BML 1996) already required the calculation of fertilizer requirements based on plant needs and soil replenishment, but it did not need to be documented, so it was not strictly adhered to. Documentation became mandatory only with the amendment of the GFO in 2017 (BMEL 2017b).

A similar challenge was the assessment of seasonal N supply from added organic matter, such as organic fertilizers or N-rich crop residues. Organic fertilization was long seen as a strategy to retard the release of plant-available N over a longer period, with higher chances of being synchronous with the crop's N demand. However, organic sources also released significant amounts of N when there was no crop present, creating "leaky spots" in the rotations where N was easily lost through leaching and volatilization (Cameron et al. 2013).

To overcome the problem of unknown mineral nitrogen release during the growing season, several additions to the N_{min} method have been suggested. One of these was the "fertilization window" (Rimpau 1984), a small plot on the field to which a reduced fertilizer dose was applied, and the greenness of the crop was observed as a proxy of its tissue N concentration, indicating N deficiency for incipient chlorosis. The fractionation of soil organic matter via electroultrafiltration (EUF) was developed as an alternative indicator of the nitrogen release capacity of the soil (Németh et al. 1987; Wiklicky 1982). However, comparisons with incubation experiments (Severin et al. 1985) and N supply on covered fallow plots (Köhler 1983) could not confirm the usefulness of this indicator. Expert systems have attempted to optimize N fertilization using multiple field experimental data, field history, management, and weather information (Schoop 1998; Schoop and Hanus 1988). Additional measurements of the crop N status during the main growth period were suggested to determine the nitrogen demand for within-season applications, for example, by analyzing the N concentration at the stem base (Wollring1983; Wollring and Wehrmann 1990) or using hand-held optical sensors (Jemison and Fox 1988; Geyer and Marschner 1990; Reeves et al. 1993; Rosso et al. 2022). Later, tractor-mounted active and passive optical sensors were developed to detect the N status of crops on the go (Leithold 2000; Reusch et al. 2002). An overview of currently available methods to estimate soil mineralization potential is given in Clément et al. (2025).

Nitrogen mineralization in the open field is strongly determined by weather conditions, essentially by temperature and humidity (Beck 1983). To predict these temporal dynamics, simulation models were developed from the 1970s onward (Beek and Frissel 1973). The first model approaches were initially very complex and attempted to represent coupled carbon and nitrogen mineralization and immobilization by explicitly simulating the population dynamics of microbial biomass in the soil (Hadas et al. 1987; Kersebaum and Richter 1994; Parnas 1975; Smith 1982; van Veen 1977). To determine the potential mineralizable nitrogen, various fractions of organic matter were used. However, the differentiation according to substance components (carbohydrates, protein, cellulose, lignin) (Beek and Frissel 1973) or substance groups (Hagin et al. 1984; Juma and Paul 1981; McGill et al. 1981; Smith 1982; van Veen et al. 1981; van Veen et al. 1984) turned out to be impractical for describing the processes in the field.

More successful for modeling was the definition of conceptual pools. Based on a kinetic approach by Stanford and Smith (1972) to describe the net mineralization of nitrogen by first-order kinetics, Molina et al. (1980) and Richter et al. (1982) were able to distinguish two fractions with different turnover coefficients in incubation experiments or three fractions when using dried soils (Richter et al. 1982). The reaction coefficients determined in the incubation experiments at different temperatures were used in models to calculate mineralization in the field using temperature functions (Kersebaum and Richter 1991; Kersebaum et al. 1986; Molina et al. 1983). Process-based models for the simulation of crop growth were developed during the 1960s by de Wit and Brouwer (1968). A first soil nitrogen model to predict crop responses under both water and N deficits was developed by Seligman and van Keulen (1981). From the end of the 1980s, several simulation models were developed combining dynamic (carbon and) nitrogen approaches for soil processes with models for crop growth (Groot et al. 1991). To represent nitrogen stress, crop models often use the concept of the critical N concentration in biomass and the nitrogen nutrition index (NNI), which is the ratio between the actual and the critical plant N concentration at a particular biomass. Since the critical plant N concentration decreases with increasing biomass, models use the so-called N dilution function (Greenwood et al. 1990; Lemaire et al. 1984) to estimate the NNI at any level of crop biomass. However, model input requirements were often too ambitious for the actual data availability at the farms. Therefore, real-time applications of models to estimate crop nitrogen demand and derive nitrogen recommendations during the growing season remain sparse (Kersebaum and Beblik 2001; Melkonian et al. 2007). Regional N budgets were combined with simulated nitrate loss in the unsaturated zone to estimate nitrate leaching at the catchment scale (Eschenbach et al. 2018; Wendland et al. 2009). Denitrification was simulated using an empirical stage model (Gäth et al. 1997), which was later refined by implementing Michaelis–Menten kinetics (Kunkel and Wendland 2006).

Soil variability is a result of different hydrological, biological, and geomorphologic processes in landscapes (Moore et al. 1991). During the 1970s and 1980s, awareness of spatial variability in soils increased (Beckett and Webster 1971; Burgess et al. 1981; Warrick and Nielsen 1980; Webster 1985). The common practice of determining the N_{min} content by means of a composite sample over the field did not represent the plant-available nitrogen at all points (Dahiya et al. 1985). Remote sensing and the technical development of harvester-mounted yield monitors linked to global navigation satellite systems allowed us to estimate spatial variation in crop growth within fields (Auernhammer et al. 1994). This led to the idea of site-specific management in the frame of Precision Agriculture (PA) (Auernhammer 1999; Stafford 2000). In practice, PA became more attractive through better agricultural equipment enabling variable-rate application of agrochemicals after 2000. Digital soil mapping has opened new opportunities to provide spatial soil information (Behrens and Scholten 2006; Michael Mertens et al. 2008) for high-resolution modeling (Kersebaum and Wallor 2023) and model-based fertilizer recommendations (Kersebaum et al. 2005; Sela et al. 2016).

6 | Outline of Measures to Mitigate Nitrogen Losses

In addition to technical solutions to adapt the nitrogen supply to the demand of plants, various management options have been proposed to reduce nitrate leaching and gaseous N emissions. These measures can be categorized into three complementary strategies: (1) reducing nitrogen availability during periods of high leaching risk, (2) retaining residual nitrogen within the soil–plant system, and (3) improving nitrogen use efficiency in general.

The first strategy aims to reduce the amount of mineral nitrogen present in the soil in autumn and winter, when crop nitrogen uptake is limited, and precipitation often exceeds evapotranspiration. Here, the N_{min} content in autumn is a reliable indicator of the potential for nitrate leaching over the winter period (Bischoff et al. 2022; Delin and Stenberg 2014; e.g., Wey et al. 2021). Measures such as avoiding autumn fertilization and reducing autumn tillage to slow the mineralization of soil organic matter can decrease the accumulation of soil mineral nitrogen by limiting external nitrogen inputs (Hansen and Djurhuus 1997; Cameron et al. 2013). Consequently, less nitrate would be available for transport downward with percolating water.

The second strategy focuses on retaining residual nitrogen within the agricultural system. Catch crops thereby play a key role in mitigating nitrate leaching by taking up mineral nitrogen remaining in the soil after harvest and temporarily storing it in their biomass. This nitrogen can become available to the next crop and should therefore be considered when calculating fertilizer recommendations (Noltemeyer et al. 2021). The effectiveness of different species of catch crops varies with sowing date and nitrogen availability. Early sowing generally enhances nitrogen retention (Noltemeyer et al. 2021; Teixeira et al. 2016). However, as discussed above, catch crops may also increase N_2O emissions under certain conditions (Helfrich et al. 2024; Nasser et al. 2025). In addition, catch crops can contribute to the build-up of soil organic matter (Liang et al. 2023), which removes carbon from the atmosphere but may increase nitrogen mineralization (Constantin et al. 2012) and, consequently, nitrate leaching and N_2O emissions (Giweta et al. 2017; van Groenigen et al. 2017).

Likewise, appropriately designed crop rotations can improve the temporal synchronization between nitrogen release and crop demand, thereby reducing periods during which excess soil mineral nitrogen is susceptible to leaching. In this context, widening narrow crop rotations may allow for the establishment of catch crops after potatoes or winter oilseed rape, which leave high amounts of residual nitrogen. In addition, the inclusion of crops with comparatively low nitrogen demand, such as sugar beet, can further contribute to reducing nitrogen surpluses and consequently N leaching within the cropping system (Rose et al. 2023; LWK Niedersachsen, LBEG 2022, LBEG 2024a).

Certain crops are associated with elevated residual soil nitrogen levels and, consequently, an increased risk of nitrate leaching. For example, intensive soil disturbance during potato harvest disrupts soil aggregates and stimulates the mineralization of soil organic matter (Neumann et al. 2012). Similarly, high N_{min} contents and subsequent NO_3 leaching have frequently been observed following winter oilseed rape, as large amounts of crop residues with a low C to N ratio remain in the field and are readily mineralized (Rose et al. 2023; LWK Niedersachsen, LBEG 2024a). In addition, crops such as silage maize, brassica vegetables, and leafy vegetables that are harvested for dark-green leaves or high turgidity often leave substantial amounts of residual N in autumn due to their high N fertilization rates and crop-specific characteristics. In contrast, crops such as sugar beet and summer malting barley generally result in lower residual soil mineral N levels because of their comparatively lower N demand and fertilizer requirements (Biernat et al. 2020; Dieser et al. 2023; Rose et al. 2023; Tei et al. 2020).

The third strategy seeks to enhance nitrogen use efficiency in general and thereby minimize nitrogen surpluses. Reduced fertilization of certain crops, such as silage maize, can substantially lower nitrate leaching while maintaining high crop yields (Noltemeyer et al. 2021; Heumann et al. 2013). Accurately estimating the availability of nitrogen from the mineralization of organic fertilizers and soil organic matter enables more precise fertilizer management (Heumann et al. 2011a, 2011b; LWK Niedersachsen, LBEG 2024a, 2024b). In addition, optimizing site conditions—including irrigation management, soil pH, and essential nutrient supply—can promote crop growth and nitrogen uptake. This further reduces the amount of residual nitrogen remaining in the soil after harvest (Bischoff et al. 2022; Cameron et al. 2013).

In addition to these management options, several technical solutions are available to reduce NH_3 and N_2O emissions. These measures are discussed in Sections 3.1 and 3.3.

At the system level, organic farming was suggested as one option to reduce N losses since the system allows only organic fertilizers and a limited import of farm external manure (EEC 1991). It is usually nitrogen-limited, with legumes and organic fertilizers as important sources of N. Appropriate management in autumn appears to allow for some control over nitrogen leaching in organic farming. (Askegaard et al. 2011). Several authors found a reduction in N leaching per area compared to conventional systems (Benoit et al. 2015; Biernat et al. 2020; e.g., Kirchmann and Bergström 2001). However, when relating nitrate leaching to the production of grain equivalents, the conventional system was better (Biernat et al. 2020; Kirchmann and Bergström 2001). In contrast, Benoit et al. (2015) found a reduction in nitrate leaching as well as N_2O emissions under organic farming per area as well as per unit of produced protein. Given the inconsistent findings across studies, a conclusive assessment of organic farming with regard to the reduction of N losses is not possible.

When establishing measures to avoid unwanted emissions in agricultural ecosystems, it is necessary to consider their impact on the entire system to prevent pollution swapping (Cameira

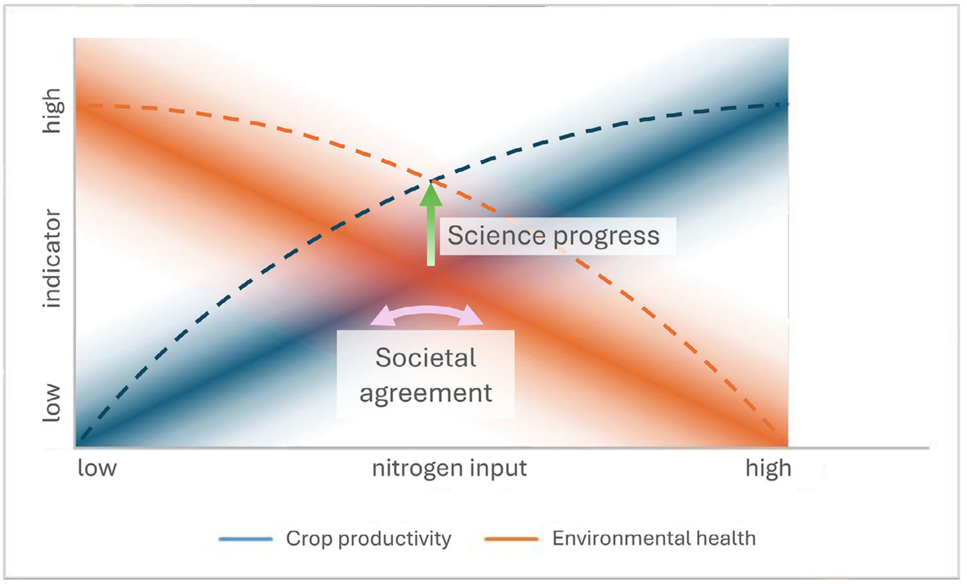

FIGURE 8 | Simplified schematic of how crop productivity and environmental health relate to nitrogen input. Whether crop production inclines more toward maximum food output or toward minimum environmental damage per invested unit of nitrogen is a societal decision. Science can help to find optima that allow higher production at lower environmental impact, for example, by improving the system's nitrogen use efficiency.

et al. 2020). For example, as discussed above, incorporation of organic fertilizers to reduce NH_3 losses can increase N_2 and N_2O fluxes (Webb et al. 2010). For measures leading to lower yields, leakage effects from the necessary replacement of yield gaps by imports must be taken into account (Agora Agriculture 2024). Part of such imports might come from countries with lower sustainability requirements. However, there are also win–win effects. For example, reducing gaseous nitrogen loss could save mineral fertilizer, which could mitigate GHG emissions from fertilizer production (Menegat et al. 2022).

7 | Ensuring Food Security and Environmental Health Simultaneously

When mineral nitrogen fertilizer is introduced into agricultural systems, emissions into adjacent environments are inevitable, as shown in the previous sections. As a result, the massive increase in food production driven by industrial agriculture (Figure 4) was associated with increasing nitrogen losses to the environment. To balance crop production and environmental health, it is key to find proficient solutions that help to maximize yield while avoiding nutrient losses into the environment as much as possible (Figure 8).

Despite longstanding awareness of these issues and extensive theoretical knowledge on environmentally sustainable nitrogen management, the translation of this knowledge into policy and practice has remained limited for many years. Numerous political declarations of intent and regulations were put into place at German and European levels to limit the nitrogen supply and enforce good agricultural practices. This led to a drop in fertilizer use during the early 1990s (Figure 3D). From the early 1990s to 2016, the N fertilizer use (mineral and organic) on cropland in Germany remained stable at approximately 170 kg N ha^{-1} (Figure 3D). Despite this, nitrogen output from harvests increased until 2017, reducing Germany's cropland N field balance from 120 kg N ha^{-1} in 1991 to 74 kg N ha^{-1} in 2017 (Figure 3D) and improving nitrogen use efficiency (NUE). In contrast, a marked intensification of livestock farming in northwestern Germany led to a regional increase in nutrient surpluses until the 2010s (Häußermann, Klement, et al. 2020).

Only following the stricter fertilization regulations in 2017 and 2020 did nitrogen input significantly drop again after having been relatively constant for more than 20 years (Figure 3C, D). In addition, the sale of mineral fertilizers has decreased in 2022 due to increased sales prices. With crop yields remaining nearly stable (DESTATIS 2025), this further declined Germany's cropland N balance to 40 kg N ha^{-1} in 2022 (Figure 3D). N field balance data from 576 practical fields on 48 farms across Germany show that the average nitrogen input decreased from 205 kg N ha^{-1} (2015–2017) to 175 kg N ha^{-1} (2019–2023) (BMEL and BMUV 2024). N-input reductions were most pronounced in silage maize, winter wheat, and oilseed rape. At the same time, nitrogen output from harvest declined only slightly from 155 to 148 kg N ha^{-1}, markedly improving NUE (N-output/N-input) in plant production from 76% to 85% (BMEL and BMUV 2024). In a meta-analysis of variety trials at multiple locations, significant reductions in nitrogen application rates of up to 28.9% between 1995 and 2021 were found (Laidig et al. 2024). Simultaneously, yields and NUE improved due to breeding advances (Figure 4B) and through better adaptation to nutrient requirements. However, the legacy of long-term intensive nitrogen fertilization will continue to supply crops with soil-borne nitrogen for at least a number of years (Sebilo et al. 2013). It remains to be seen whether a decline in yields will be delayed under enduring low fertilizer application rates. Higher average temperatures and more pronounced wet–dry cycle fluctuations, as expected in the future, might further enhance mineralization (Manzoni et al. 2020).

For a more targeted evaluation of the effectiveness of the mitigation measures, they need to be monitored in a timely manner and at the scale of application of these measures, that is, at the field level (Zieseniß et al. 2026). In light of the aforementioned challenges and in response to the EU Commission's urging, Germany has proceeded with the implementation of an enhanced monitoring framework, encompassing extensive data collection of nitrogen management on practical farms within designated model regions (Ziesenіß et al. 2026) and detailed modeling of the water and nitrogen balances at the national scale (Zinnbauer et al. 2024).

Another option to determine the effectiveness of mitigation measures is to establish an age-dependent evaluation of the groundwater measuring points, as successfully implemented in Denmark (Hansen et al. 2017). Otherwise, decreasing nitrate trends at monitoring sites with younger groundwater ages may be overshadowed by sites exhibiting nitrate loads from previous decades.

It seems that Germany has reached a stage where further nitrogen cuts in cereals might lower yields and quality (LWK Niedersachsen and LBEG 2024a, Noltemeyer et al., 2020). However, available data do not allow for the clarification of the role of N in the current stagnation of crop yields. While the unfavorable weather conditions of recent years may partly explain the recent yield stagnation (Klages et al. 2020; e.g., Riedesel et al. 2024), there may have been years during this period with more favorable conditions in which crop yields were already limited by nitrogen deficiency, preventing them from reaching their full potential (Martre et al. 2024). In Denmark, the obligation to reduce N fertilization by 15% below the economic optimum has already resulted in baking qualities for wheat no longer being achieved (Styczen et al. 2020a, 2020b). Therefore, this measure was abolished again in 2016 (Ortmeyer et al. 2023). In addition, France has observed a similar decline in baking quality since 2016 (FranceAgriMer and ARVALIS 2024). In Germany's NVZs, a restriction on the use of nitrogen fertilizer to 80% of the required amount since 2021 has resulted in relatively small yield losses of approximately 5% for wheat and barley and no significant yield loss for maize (Mittermayer et al. 2024). However, it led to a significant decrease in grain protein (Mittermayer et al. 2024). Furthermore, when considering lower production in Germany, the potential leakage effects of importing products from other countries where production may be associated with higher or lower emissions must be considered (Agora Agriculture 2024), as well as the possible impact on global food security (Burney 2025). Finally, several scientific studies suggest that reducing mineral nitrogen fertilization below the economic optimum does not necessarily reduce nitrogen leaching (Delin and Stenberg 2014; Kage et al. 2022; Lord and Mitchell 1998).

From this perspective, further reducing nitrogen inputs in crop production would most likely not achieve the desired outcome. A potential lever emerges when we consider the spatial distribution of nitrogen surpluses, which are particularly high in regions where intensive livestock farming takes place (Häußermann, Klement, et al. 2020). In addition to suggested measures to mitigate emissions from livestock systems themselves (Epper et al. 2025), reducing the animal stocking density and transferring excess animal excreta as fertilizers to neighboring arable farming regions, as is already practiced, could further reduce nitrogen emissions from the larger agricultural system (Häußermann, Klement, et al. 2020). This is already supported by an upper limit of 170 kg N ha^{-1} for organic fertilizer application (BMEL 2020). Slurry separation that produces a solid phase that still contains up to 25% of the original slurry's nitrogen but helps to reduce transport costs (Møller et al. 2000), which could further motivate such regional trade. Taking this idea further, there is potential for synergy in reducing livestock production to support climate change mitigation goals (by reducing methane emissions) and increasing food security (by producing more efficient plant-based food on sites where crop production is possible) (Chai et al. 2019; Fresán and Sabaté 2019; Springmann et al. 2018). If people gradually shift toward a more plant-based diet, supported by appropriate policy measures, a reduction in meat and milk production in Germany would help to control the distribution of nitrogen on farmland more effectively, likely resulting in further reductions in nitrogen emissions (Springmann 2023).

8 | Conclusions

In developing strategies for research and policy on efficient N use in agriculture, we have demonstrated that balancing its beneficial and harmful effects creates fundamental tensions. The most technologically advanced agricultural societies, which are also the most committed to environmental protection, now face a paradox: any effort to improve one aspect risks undermining the other. While it is imaginable that German society would accept lower crop production if it led to less pressure on the environment, individual farmers probably would not if they were not properly compensated for their economic loss. The argument that the baking quality of wheat is declining is perhaps not as robust as it may initially appear. Obviously, there are many ways to produce high-quality bread and pastries using lower-protein flours (Chakrabarti-Bell et al. 2017; Dapčević-Hadnađev et al. 2022). On the other hand, if the potential to further reduce emissions into the environment by applying less-than-optimal nitrogen to cereals is almost nonexistent, it seems not worth thinking further in this direction. The story is slightly different from that of vegetables, where the supply of nitrogen is directly related to customer choices. Here, education at the consumer end of the value chain could help reduce the pressure on farmers to apply high doses of nitrogen to ensure dark-green colors and crispiness of their produce. Currently, vegetable-growing areas in Germany seem not very relevant in terms of their land-use share (0.8% in 2025; Statistisches Bundesamt 2026), but they may still be responsible for local nitrate pollution. However, this consideration shows that there is more to the black-and-white picture of productivity versus nitrogen inputs, and NUE is key. A proper design of the crop rotation with a mix of cereals and careful autumn management of the crop residues in the field are long-existing recommendations to reduce nitrogen emissions from field vegetable production. In addition, for arable farming, the rotation design, including residue management and the use of intermediate crops—particularly catch crops—is an important lever to further reduce N emissions. Further potential measures to improve the NUE in cropping systems have been discussed, and they need to be further optimized, particularly specific to site and weather conditions.

Even more relevant are the emissions related to animal production and the various ways they release unproductive N into the environment. Our discussion has identified the greatest potential for reducing N emissions in this area. While we have mainly discussed mitigation measures associated with slurry application on fields, additional measures are implemented directly in the barn, such as demand-driven feeding, immediate separation of urine and manure, emission-reducing floors and gutters with closed slurry channels, and chemical or biological air scrubbers in the ventilation system. These measures can be implemented without reducing current levels of milk and meat production, but reducing these levels could contribute even more to achieving synergies with climate change mitigation and global food security goals.

However, two major uncertainties remain. First, will the balance between production and environmental protection goals be disturbed and the expected efficiency of NUE-improving measures be reduced when climate change is aggravated? Second, how would the entire equation change if a definitive causal link were confirmed between the continuous intake of low concentrations of nitrate and the development of cancer? The current discussion on this matter in Denmark (Heissel 2026) illustrates how science must remain vigilant and continually assess environmental processes and associated control mechanisms to inform future political decisions.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

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

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