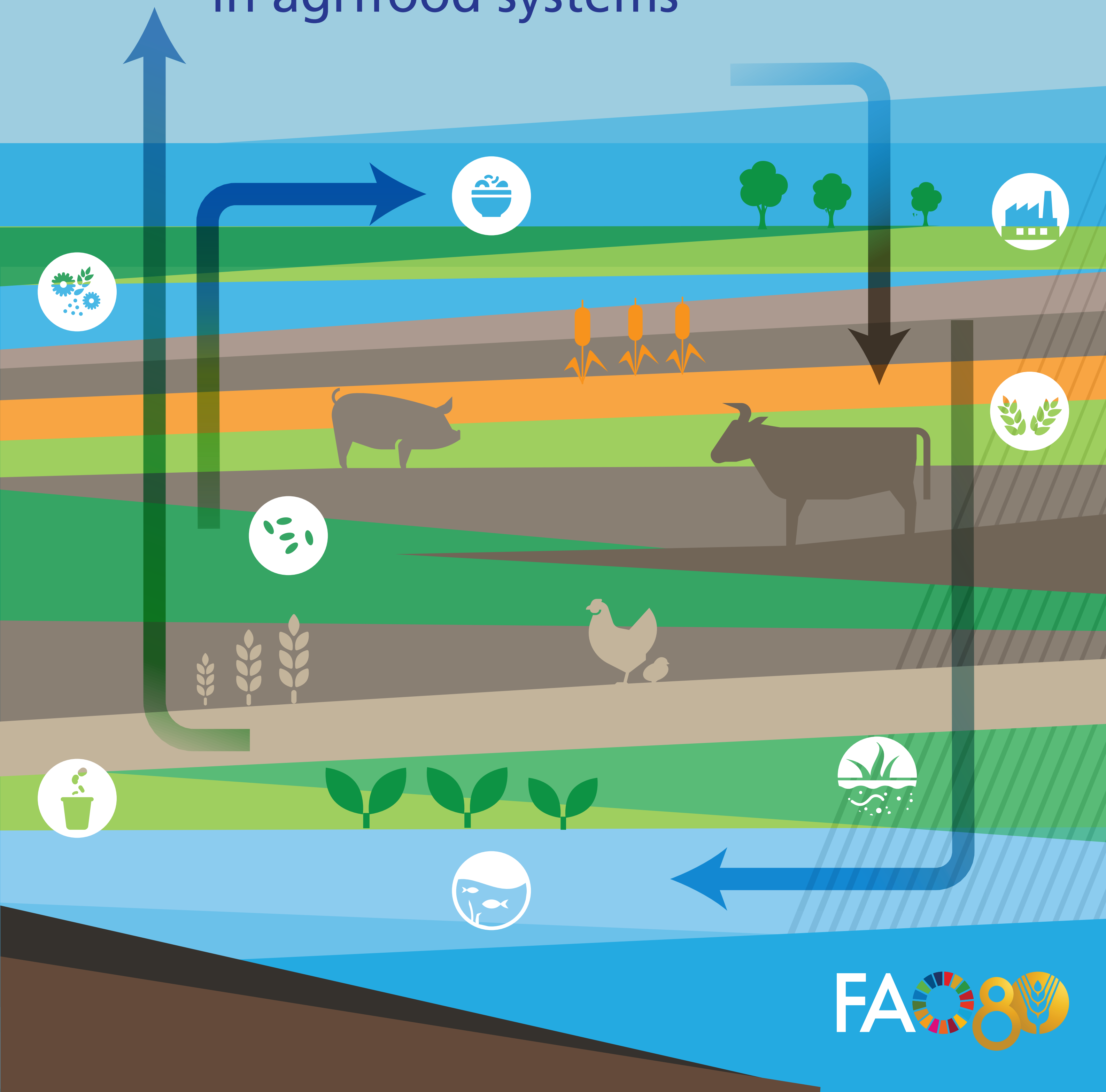
Food and Agriculture
Organization of the
United Nations

# Sustainable nitrogen management in agrifood systems



# Sustainable nitrogen management in agrifood systems

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# Preface

Nitrogen is a vital element for the natural environment and forms essential compounds for all living organisms. As it cycles through air, soil and water, nitrogen plays a crucial role in the functioning of ecosystems. The growth of agrifood systems has relied on the access to nitrogen resources from both industrial sources and biological fixation. The use of these nitrogen sources has increased agricultural production and has contributed to food security for a growing world population. If not managed properly, overuse of nitrogen affects air, water and soil quality. In contrast, mining nitrogen from soils leads to soil degradation, thus driving biodiversity loss and exacerbating climate change. Agrifood systems are a main driver of disrupted nitrogen cycles at national, regional and global levels, and contribute to the negative impacts on the environment. To ensure the sustainable use of nitrogen as a vital natural resource, all stakeholders in agrifood systems should take action to adopt practices that enhance nitrogen use efficiency and minimize pollution and waste.

This report gives a comprehensive overview of the role of nitrogen use and consequent challenges in agrifood systems. It offers solutions for crop and livestock systems on how to improve nitrogen management to enhance productivity and outlines the potential of adopting circular bioeconomy approaches to enhance nitrogen use efficiency and minimize pollution. Through this, a transformation in agrifood systems where nitrogen use is balanced can ensure food security, nutrition and farmers' livelihoods.

A transformation of the agrifood system to enhance the three dimensions of sustainability necessitates collaborative efforts from all stakeholders, from local to global levels. This report underlines the importance of policies promoting sustainable nitrogen management and how national commitments can reduce nitrogen pollution. By joining our efforts to reduce nitrogen pollution and waste, agrifood systems contribute to the achievement of the Paris Agreement and the Sustainable Development Goals, in particular, SDGs 2, 6, 12, 13, 14, 15 and 17.

As such, this report is the first FAO publication on sustainable nitrogen management in agrifood systems and how to ensure the use of this vital resource now and in the future for improved production, nutrition, a healthier environment, and a better quality of life. It is hoped that the results and recommendations of this report bolster efforts of countries and agrifood system stakeholders to commit to addressing nitrogen challenges while transforming agrifood systems through better production, better nutrition, better environment, and a better life for all, leaving no one behind.

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# Abbreviations

| | |
|---|---|
| **4Rs** | nutrient application with right source, right rate, right time and right place |
| **CAN** | calcium ammonium nitrate |
| **COS** | characteristic operating space |
| **CS** | cattle slurry |
| **DM** | dry matter |
| **FAO** | Food and Agriculture Organization of the United Nations |
| **FCR** | feed conversion ratio |
| **FLW** | food losses and waste |
| **GDP** | gross domestic product |
| **GHG** | greenhouse gas |
| **GLEAM** | Global Livestock Environmental Assessment Model |
| **GTP** | global temperature change |
| **GWP** | global warming potential |
| **IPCC** | Intergovernmental Panel on Climate Change |
| **ISP** | input subsidy programme |
| **LAC** | Latin American and the Caribbean |
| **LEAP** | Livestock Environmental Assessment and Performance |
| **LCA** | life cycle assessment |
| **LCI** | life cycle inventory |
| **LCIA** | life cycle impact assessment |
| **LMICs** | low- and middle-income countries |
| **NUE** | nitrogen use efficiency |
| **SDG** | Sustainable Development Goal |
| **SOC** | soil organic carbon |
| **SOM** | soil organic matter |
| **TASF** | terrestrial animal-sourced food |
| **TMP** | technologies and management practices |
| **USD** | United States dollar |

## UNITS AND CHEMICAL FORMULAS

| | |
|---|---|
| **Al** | aluminium |
| **$Al^{3+}$** | aluminium ion |
| **C** | carbon |
| **°C** | degrees Celsius |
| **$CH_4$** | methane |
| **$CO_2$** | carbon dioxide |
| **$CO_2eq$** | carbon dioxide equivalent |
| **Gt** | gigatonne, metric unit equivalent to 1 billion ($10^9$) tonnes |
| **$H^+$** | hydrogen ion |
| **ha** | hectare |
| **K** | potassium |
| **kg** | kilogram, a unit of mass equal to 1000 grams |
| **mg** | milligram |
| **N** | nitrogen |
| **$N_2$** | dinitrogen |
| **NBPT** | N-(n-butyl)thiophosphoric triamide |
| **$NH_3$** | ammonia |
| **$NH_4^+$** | ammonium |
| **$N_2O$** | nitrous oxide |
| **$NO_2^-$** | nitrite |
| **$NO_3^-$** | nitrate |
| **$NO_x$** | nitrogen oxides |
| **$O_3$** | ozone |
| **$OH^-$** | hydroxyl radicals |
| **P** | phosphorus |
| **pH** | quantitative measure of hydrogen ion concentration |
| **Tg** | teragram, metric unit equivalent to 1 million ($10^6$) tonnes; equivalent to a megatonne |
| **yr** | year |

# Executive summary

Nitrogen (N) is one of life's fundamental building blocks – a core ingredient in amino acids and proteins – and essential for agrifood systems. With the invention of the Haber–Bosch process, humans have converted unreactive $N_2$ to reactive forms of N that can be used as mineral fertilizer, which has significantly contributed to increased crop production and yields to feed a growing world population. This technological breakthrough has altered the global N cycle, resulting in excess N release to the environment, negatively impacting air and water quality, human health and biodiversity. The anthropogenic release of N impacts both terrestrial and aquatic ecosystems, as altered N flows have profound effects on natural ecosystem structure, function and services upon which humanity depends. Nitrogen losses occur through emissions of ammonia ($NH_3$) and nitrogen oxides ($NO_x$), which lead to air pollution, and nitrous oxide ($N_2O$), a potent greenhouse gas (GHG), which contributes to climate change. Additionally, N can be lost through nitrate ($NO_3^-$) leaching in soil and water bodies. This causes eutrophication and acidification, ultimately harming terrestrial and aquatic ecosystems, contributing to biodiversity loss, and affecting the provision of clean air and water. As a result, the substantial losses of anthropogenic N pose a risk to human health and contribute to the triple planetary crisis of climate change, pollution and biodiversity loss.

Agrifood systems play a significant role in the alteration of N dynamics. Nitrogen is an essential building block for crop and livestock production. While some plants, such as legumes, can access atmospheric N through biological N fixation, most others depend on N availability in soils. Synthetic N fertilizer has complemented natural processes and significantly increased crop yields. The over-application of synthetic N fertilizer has resulted in substantial losses of N through the leaching of nitrates when the absorption capacity of soils is surpassed. Additionally, with the rising demand for livestock products, the livestock sector has undergone significant changes, transitioning from traditional and small-scale to intensive production systems in which large amounts of concentrated feeds are used. Production of this feed is partly linked to deforestation and heavy synthetic fertilizer use, causing gaseous N emissions and $NO_3^-$ leaching. As feed production is typically decoupled from livestock-dense production areas, high concentrations of manure are accumulated in the latter areas. Significant emissions occur in livestock-dense housing systems where manure is accumulated and stored for prolonged periods, causing emissions of $NH_3$, $N_2O$ and $NO_x$. The livestock sector is the main contributor of N losses by agriculture and represents about one-third of total N emissions from anthropogenic activities.

Conversely, many low- and middle-income countries still face opposite challenges, as N fertilizers are more difficult to access. Here, soil health degradation occurs when crops are harvested without compensating for the harvested N through the application of organic (such as manure, compost and crop residues) or synthetic fertilizer, which in turn results in crop yields well below their potential. Additionally, manure is often not collected or correctly handled, resulting in emissions of N and loss of nutrients that could otherwise be returned to the agricultural system.

Because of its significant role in environmental N pollution, the agricultural supply chain must implement sustainable N management practices to minimize N losses and increase balanced N cycling within the agrifood system. Sustainable N management is defined as management practices that seek to minimize external N inputs and losses and increase recycling of N within the production system. Increasing N use efficiency (NUE) can contribute to sustainable N management. Nitrogen use efficiency is the ratio of N recovered in the final output to the total N used as input. Increasing NUE aims to recover as much as possible of the N input as possible in the final product, thereby minimizing the amount of N lost in the production process. Improved fertilization strategies contribute to improving NUE and sustainable N management in cropping systems. In livestock production, strategies at the farm level to increase NUE should focus on minimizing N excretion through manure. When feeding high-protein diets, a significant amount of N is excreted via manure (faeces and urine). Improved feeding strategies, including low-protein feed, can decrease manure N excretion and associated N losses. Through improved manure handling and storage and an adequate use of manure N in crop production, overall NUE can be increased substantially. Reducing N losses from manure can be achieved through innovative livestock housing systems, improved storage, and low-emission application of manure to cropland. Beyond farm-level measures, crop–livestock integration and recoupling of livestock to local feed production can enhance sustainable N management on a regional scale. In general, livestock decrease the overall NUE of the food production system compared with plant-based production systems, as they add additional steps in the process where N can be lost to the environment. Integrating livestock systems with crop production systems improves NUE of the system as a whole, thereby contributing to an increase in sustainable resource use.

Improving resource use efficiency, including NUE, can be achieved by the adoption of circular bioeconomy principles. The circular bioeconomy aims to provide sustainable solutions in the production, utilization, conservation and regeneration of biological resources within and across all economic sectors to enable a transformation to a more sustainable economy. Within agricultural production systems, circularity principles are proposed to improve resource use efficiency and NUE and can contribute to sustainable N management as they aim to maximize the efficiency with which food is produced and utilized. The main principles of a circular agrifood system are to reduce food losses and waste, recycle inevitable food losses and waste back into the agrifood production chain, use arable land primarily for direct human consumption to maximize resource use efficiency and available food, and use livestock to convert biomass and waste streams unsuitable for human consumption. Across these principles, many solutions are present to increase the recycling of N and increase NUE of the agrifood system. In recognition of the importance of these processes, countries can now use NUE as one of the indicators of the productivity and sustainability of their agrifood systems when reporting on Sustainable Development Goal (SDG) Indicator 2.4.1.

Nitrogen management policies in agrifood systems present disparities across different regions. Policies often prioritize food security and productivity gains, leading to high N inputs and low NUE. For instance, Asia's Green Revolution saw significant crop yield increases as a result of fertilizer subsidies. These policies have led to massive environmental pollution from the overuse of synthetic fertilizers. In response, Asian countries have implemented reforms to reduce N fertilizer use and improve NUE. Africa has challenges related to low crop yields and soil nutrient depletion due to inadequate policies, low-fertility soils, and limited access to affordable synthetic fertilizers. The European Union and North America have achieved higher NUE through nutrient management guidelines and environmental regulations. Conversely, Latin America and the Caribbean face challenges due to heavy reliance on imported fertilizers, which are affected by fluctuating prices and disruptions of supply chains.

Adoption of agricultural technologies and financial mechanisms, such as crop insurance and joint responsibilities among agrifood chain stakeholders to decrease N loss and share abatement costs are needed. Countries are encouraged to adopt policies that promote sustainable N management and address other environmental challenges, such as climate change, water use and biodiversity loss.

Sustainable N management is crucial for achieving the SDGs by 2030, particularly those related to ending hunger (SDG 2), health (SDG 3), clean water (SDG 6), sustainable production and consumption (SDG 12), climate action (SDG 13), and preserving life underwater (SDG 14) and on land (SDG 15). In developing countries, improving NUE can improve soil health and fertility, increase crop production and yields, and increase food production. Furthermore, improving NUE can contribute to improved human and environmental health by reducing harmful emissions and protecting water bodies from pollution. For these efforts to be successful, policies need to reconcile the dual role of N as an important nutrient necessary for economic growth, human advancement and food security and as a pollutant that causes serious ecosystem damage.

Key actions and policy options to promote sustainable nitrogen management should focus on the following.

- *Improve nitrogen management in crop production* through promoting the use of biological N fixation in locally suitable crop rotations and encouraging the use of manure as organic fertilizer. Low- and middle-income countries should enhance access to synthetic fertilizers and promote agroecological practices, while the fertilizer industry should take urgent action to cut GHG emissions during the production of synthetic fertilizers.
- *Improve nitrogen management in the livestock sector* through developing guidelines to adopt best practices in manure management and processing techniques, enhance spatial integration of crop and livestock production, and implement circular bioeconomy principles at the landscape level. Livestock farmers should improve feed formulation to optimize protein intake and improve feed use efficiency. Agrifood system policies should focus on improving spatial planning and reducing livestock numbers in areas with high geographical concentration to ensure crop and livestock systems are balanced and integrated.
- *Reduce nitrogen loss and waste.* Countries should bolster efforts to reduce food loss and waste throughout the agrifood production chain and promote recycling and treatment of food unsuitable for human consumption as livestock feed.
- *Promote public and private investment.* National governments, the private sector, international financial institutions, and local agricultural banks should mainstream sustainable N management into development projects and programmes in agrifood systems and promote investment in high-efficiency, low-emission mineral fertilizers and production of organic residues to enhance system efficiency and reduce waste of resources and environmental pollution. Agrifood system stakeholders should promote investment in agroecology and sustainable crop–livestock integrated development projects to enhance sustainable N management.
- *Capacity building at scale.* Countries and international development partners should support national capacity building on sustainable N management among different agrifood system stakeholders, including the public, private sector, civil society organizations, farmers, and producer organizations; strengthen national extension services, research, and knowledge transfer; and promote sustainable N management practices through farmer field schools, and low-N footprint diets.
- *Policy options.* Countries should promote the integration of sustainable N management in nationally appropriate mitigation actions and nationally determined contributions, including targets to reduce $N_2O$ from agrifood systems to keep the Paris Agreement goal of 1.5 °C in sight. Countries should set national commitments to reduce N pollution, including $NH_3$ and $NO_x$ emissions to air and $NO_3^-$ losses to water, in line with Target 7 of the Kunming-Montreal Global Biodiversity Framework and SDGs 6, 12, 13, 14, 15 and 17. Finally, countries should address consumption patterns and promote healthy diets with low environmental impact.

# Chapter 1
# Introduction

Nitrogen (N) is one of life's fundamental building blocks. Humans have heavily altered the global cycle of N to increase food production to satisfy the needs of a growing population (Tian *et al.*, 2022). As a result, excess N has entered natural ecosystems and contributed to the reduction of air and water quality worldwide. Through agriculture and industry, each year, humans are now adding around 150 teragrams (Tg) of reactive N (which includes all compounds of N following the fixation of atmospheric dinitrogen $N_2$) to the Earth's land surface, which is more than double the rate of pre-industrial terrestrial N fixation (Schlesinger, 2009). Projected changes in climate are expected to further increase biological and anthropogenic N fixation, potentially reaching a total of around 600 Tg N/yr by 2100 (Fowler *et al.*, 2015).

The anthropogenic release of N to the environment affects both terrestrial and aquatic ecosystems, as altered N flows have profound effects on natural ecosystem structure, function and services upon which humanity depends. According to Steffen *et al.* (2015) and Richardson *et al.* (2023), global N flows have already surpassed the planetary boundaries – a term referring to the environmental limits within which humanity can safely operate. The degree of this exceedance has dramatically increased since 2015 (Richardson *et al.*, 2023). Emissions of ammonia ($NH_3$) and nitrogen oxides ($NO_x$) have led to air pollution, nitrate ($NO_3^-$) loads in water bodies have caused eutrophication and harmed aquatic ecosystems and biodiversity, and emissions of nitrous oxide ($N_2O$), a potent greenhouse gas (GHG), contribute to climate change and $O_3$ depletion. This has weakened ecosystem resilience and reduced the provisioning of clean air and water, recreation, fisheries, forest products and biodiversity both qualitatively and quantitatively.

The assessment of N effects on the environment is challenging due to the complex nature of N dynamics. Nitrogen cycles through various oxidized and reduced forms via biological and chemical processes, allowing a single emitted N molecule to initiate a series of effects – both positive and negative – known as the N cascade (Galloway *et al.*, 2003). Because climate alters N dynamics, climate-change-induced extreme variations in temperature and precipitation are likely to increasingly weaken ecosystem resilience and alter ecosystem responses to N. Moreover, the induced effects on ecosystems can exacerbate climate change, creating a positive feedback loop – for example, N dynamics altered by climate change affect ecosystem processes, which in turn influence climate change. Thus, N pollution hinders human efforts to stay within or return to the planetary boundaries of climate change and potentially alters other boundaries, such as stratospheric $O_3$ depletion caused by catalytic cycles by N oxides (Chipperfield and Bekki, 2024).

Agriculture plays a significant role in the alteration of N dynamics. Nitrogen is essential for crop and livestock production. While some plants, such as legumes, can access atmospheric N through biological N fixation, most others depend on N availability in soils. The use of synthetic N fertilizer has complemented natural processes to significantly increase crop yields. As N fertilizer application increased with the growing demand for food and feed, losses of N to the environment increased. At the other end of the spectrum, many low-income countries still face challenges in accessing N fertilizers, leading to soil health degradation and crop yields well below their potential. Nitrogen losses in livestock systems occur indirectly (e.g. through feed production) and directly (e.g. from manure); livestock farming is the main contributor of N losses from the agrifood system. These losses contribute to environmental impacts such as air and water pollution, eutrophication, acidification, biodiversity loss and climate change (Galloway *et al.*, 2010; Leip *et al.*, 2015; Otte, Pica-Ciamarra and Morzaria, 2019; Steinfeld *et al.*, 2006; Sutton *et al.*, 2013). Sustainable N management, which focuses on minimizing external N inputs and N losses and/or maximizing N recycling, is vital for the transformation of agrifood systems in which natural resources need to be used sustainably, not exceeding planetary boundaries.

While the exceedance of the global N planetary boundaries is evident, regional heterogeneity needs to be considered, which requires different formulations of policies and improvement options suitable for each set of local conditions. For instance, countries with limited access to N fertilizers, such as those in sub-Saharan Africa and some in Latin America and the Caribbean, and Asia, currently maintain N levels within the "safe" operating zone. In these regions, a lack of nutrients in the soil often means that nutrient mining by agricultural practices contributes to land degradation, which reduces soil quality and undermines agricultural productivity. In contrast, European countries and some Asian countries have seen intense N-related pollution because of the high use and misuse of N fertilizers and manure. These regional disparities pose challenges to farmers and policymakers – the latter need to develop N policy recommendations tailored to their specific political, geographic and climatic conditions while still tackling the N challenges at a global scale. Improving N management in the agrifood system is crucial to reducing agriculture's contribution to the triple planetary crisis: climate change, pollution and biodiversity loss (UNFCCC, 2022).

This report outlines the challenges and opportunities of improving N management within the agrifood system and provides information on practical interventions aimed at enhancing the sustainable N management for FAO Members and other agrifood system stakeholders, including the private sector, non-governmental organizations, civil society organizations and farmers' organizations. Sustainable N management is defined as management practices that seek to minimize external N inputs and losses and/or increase recycling of N in the production system. The report emphasizes farm-specific solutions, provides case studies that focus on how to improve N use efficiency (NUE) and outlines the implications for the global agrifood system when implementing sustainable N management practices. Chapters 2 and 3 outline the challenges and opportunities of N use in crop and livestock systems, respectively. Chapter 4 comprehensively outlines the environmental impacts of excessive N use and N mining and how this impacts global ecosystems. Chapter 5 focuses on the circular bioeconomy, how circular agrifood systems improve NUE, and which N management practices contribute to a circular bioeconomy. Lastly, Chapter 6 provides an overview of current policies aimed at N management in agriculture and an outlook on what future policies are needed to enhance sustainable N management within the global agrifood systems.

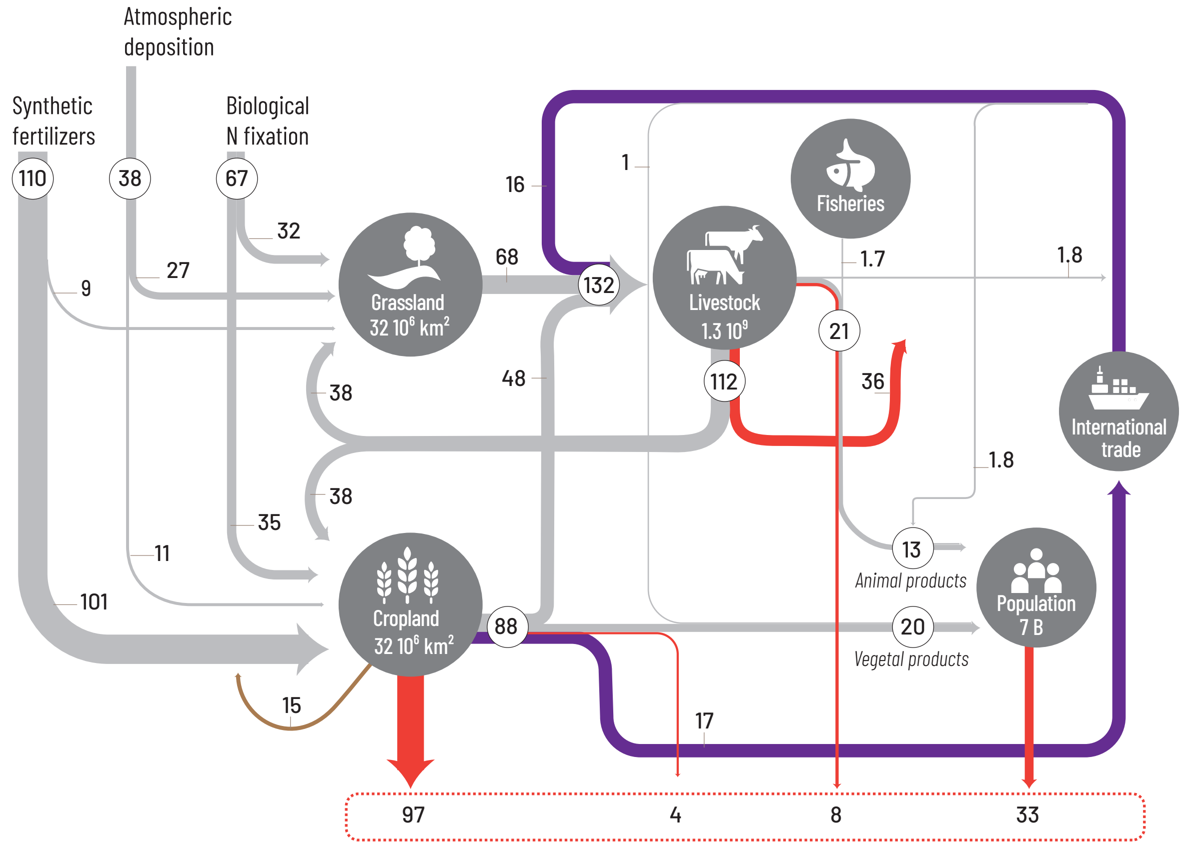
# Chapter 2
# Nitrogen use efficiency in cropping systems

## 2.1 INTRODUCTION

Nitrogen is an essential resource in agrifood systems that requires careful management to avoid environmental pollution, climate change, impacts on human health, and N misuse (Einarsson *et al.*, 2024). Nitrogen is a macronutrient and a critical component of food constituents, especially amino acids and proteins required for plant, animal and human growth (de Vries *et al.*, 2024; Sutton *et al.*, 2013). Judicious use of N in agriculture helps to avoid soil degradation and nutrient depletion and increases crop yields (FAO, 2019). Excess emissions of N into the environment from agricultural operations have damaged environmental and human health, including exacerbating global warming, degrading air and water quality, and depleting stratospheric $O_3$.

Cropping systems represent the entry point for most of the new N inputs into the agrifood system and the largest losses to the environment (Figure 1). The proportion of N inputs retained in agricultural outputs is defined as the N use efficiency (see section 2.2). Improving NUE in cropland is essential to enhancing agricultural and environmental outcomes (Bouwman *et al.*, 2013; Schulte-Uebbing and de Vries, 2021). FAO has developed the global reference

FIGURE 1
**A schematic representation of N transfers (Tg N/yr) through the global agrifood systems in 2015**




*Note:* The purple arrows represent N flows in international trade; the red arrows represent losses of N into the environment from the agrifood supply chain; fisheries includes aquaculture.

*Source:* Authors' elaboration based on **Lassaletta, L., Billen, G., Garnier, J., Bouwman, L., Velazquez, E., Mueller, N.D. & Gerber, J.S.** 2016. Nitrogen use in the global food system: past trends and future trajectories of agronomic performance, pollution, trade, and dietary demand. *Environmental Research Letters*, 11(9): 095007. https://doi.org/10.1088/1748-9326/11/9/095007 data on cropland nutrient balances, jointly with a large international community of experts from both private and public sectors to report on different indicators, including Sustainable Development Goal (SDG) 2.4.1 (FAO, 2023a; Ludemann *et al.*, 2024).

In 2022, the largest external N inputs to croplands at global level came from synthetic fertilizers (102 Tg N/yr), followed by symbiotic N fixation associated with legume crops (40 Tg N/yr) and atmospheric deposition (16 Tg N/yr). Croplands received a substantial amount of recycled N from livestock manure, applied directly on soil as fertilizer (25 Tg N/yr). A larger amount was left on pastures, grasslands and rangelands (92 Tg N/yr) (FAO, 2023a). Inputs to both croplands and grasslands have grown considerably over time (Lassaletta *et al.*, 2016; Xu *et al.*, 2019).

Nitrogen management, including all practices related to N inputs, N recycling, and avoiding N losses across global croplands, is complex and heterogeneous, with numerous opportunities to improve NUE, improve crop production, and decrease environmental degradation. The agronomic and environmental outcomes of cropping systems can differ substantially depending on how N is managed at the field level, including decisions about fertilizer placement, source, application rate and timing. Nutrient application with right source, right rate, right time and right place (the 4Rs of nutrient stewardship) optimize these management practices, and are discussed further in section 2.4. Crop rotation, including leguminous species, influences nutrient cycling (Anglade *et al.*, 2015), as do multi-purpose production systems such as agroforestry, agropastoral, silvopastoral and agrosilvopastoral systems. Furthermore, integration between crop and livestock systems can enhance N recycling and decrease the need for external inputs (Lassaletta *et al.*, 2024). Yield gaps across the world are related to the unequal distribution of N resources (Mueller *et al.*, 2012; Sinclair and Rufty, 2012; Aramburu-Merlos *et al.*, 2024); for example, the use of N inputs in Southeast Asia is ten times greater than in many low-income countries due to the difference in cropping systems and access to mineral N fertilizer (Zhang *et al.*, 2021). Decreasing yield gaps requires addressing a range of factors that define, limit and reduce yield in conjunction with N, including issues associated with crop varieties and crop rotation, soil health and fertility, water management, weed management, pests and diseases (Gerber *et al.*, 2024; Waddington *et al.*, 2010).

Improving the balance between the benefits and costs of N is possible; here, the aim is to provide recent scientific information that can enable improvements in cropland N management. First, the following sections examine how NUE is defined and used in cropping system management. Second, they outline patterns and trends in NUE across the world. Third, they identify options to improve NUE across different scales of management.

## 2.2 DEFINING NITROGEN USE EFFICIENCY IN CROPPING SYSTEMS

Nitrogen use efficiency in agriculture is an indicator defined as the proportion of N inputs that are retained in agricultural outputs in a system (Watson and Atkinson, 1999). This section focuses on the crop system (or the plant-system scale), although NUE can be estimated at other system levels (Zhang *et al.*, 2020); for example, the animal–plant–soil, agrifood (including human consumption) and landscape (including natural areas and industry) systems. Likewise, NUE can be estimated at different spatial scales, for example, at the plot (Guardia *et al.*, 2021), farm (Quemada *et al.*, 2020a), watershed (Compton *et al.*, 2021), country (Zhang *et al.*, 2021) and global (Lassaletta *et al.*, 2014b; Ludemann *et al.*, 2024) spatial scales.

Improving NUE at the cropping system scale (such as enhancing the fraction of the nutrient that is taken up by the crop) can be a win–win for agronomic, economic and environmental outcomes. Increasing NUE can improve the cost–benefit balance at the farm scale through the rational use of farm inputs, such as synthetic fertilizers (subject to price fluctuations), and using alternative N sources. Increasing NUE can reduce the environmental problems associated with losses of N to the environment. In the face of the current fertilizer crisis, improving NUE is a valuable tool to cope with the reduced fertilizer accessibility and the 2021–2022 fertilizer price spikes (FAO, 2023e).

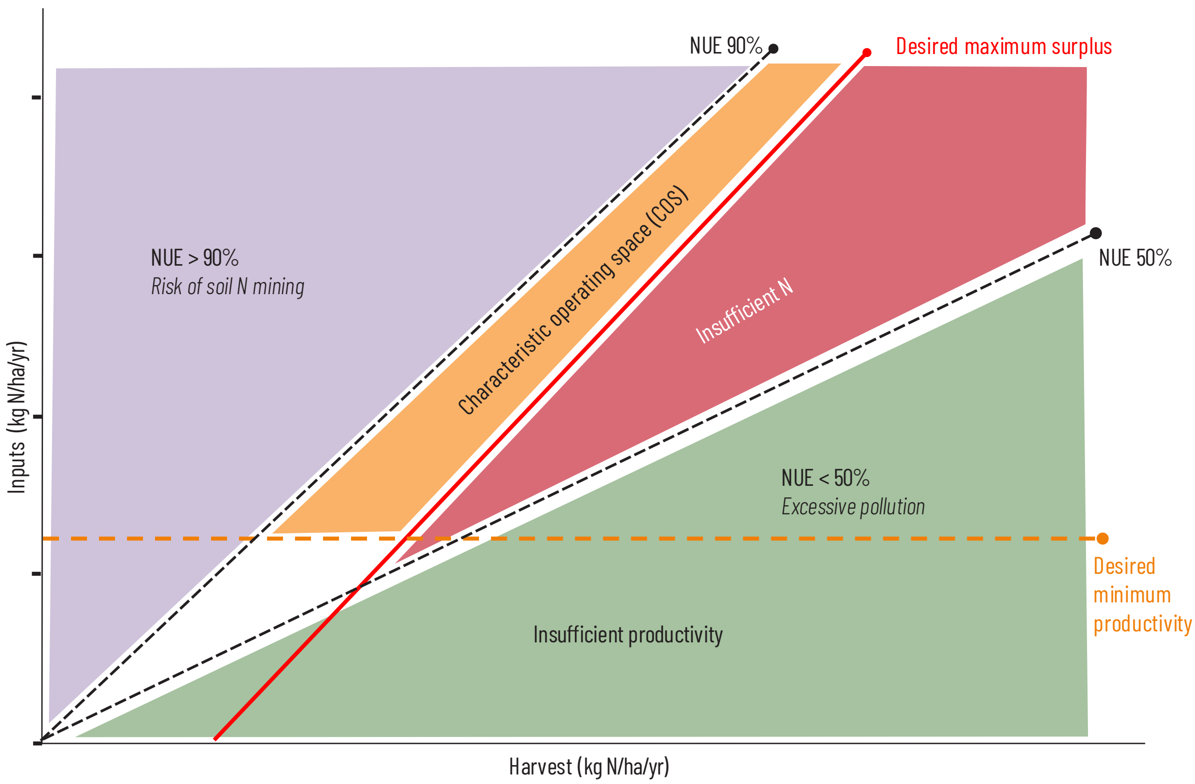
In cropping systems, the inputs comprise synthetic N fertilizer, manure N, biological N fixation, atmospheric N deposition, and N from crop residues derived from other (internal or external) systems. Additional N inputs, such as N in seeds and irrigation water and N from net mineralization of soil organic matter (SOM), should also be considered. In addition to the N in crop harvest, there are possible N losses via volatilization, leaching, runoff and soil erosion.

Soils can be a significant source of N as they can store 20 to 100 times more N than the N content in crops. In particular, SOM contains up to 5 percent N (Weil and Brady, 2017). Net N mineralization or sequestration in the SOM is not commonly evaluated, although it can be a substantial component of the budget, particularly when estimating NUE in one season only (Martinez-Feria *et al.*, 2018). To solve this issue, NUE should be defined at the scale of the full crop rotation cycle, as a given crop can use soil N resources left over by the preceding crop in the rotation. Thus, soil N is considered a source or a sink within the cropping system; neglecting the accumulation or depletion of soil N within a system will likely bias the assessment of the system NUE (Billen *et al.*, forthcoming). In soils reaching a steady state, N change could be considered negligible (van Grinsven *et al.*, 2022) except for soils in which the soil organic N content is in the process of evolution (Serra *et al.*, forthcoming). For example, soils rich in organic matter may release a substantial amount of N every year through soil cultivation practices.

There are many ways to estimate NUE. Agronomic approaches to NUE include agronomic efficiency, the ratio between yield increase (over an unfertilized control) and N applied as fertilizer; crop recovery efficiency, the increase in N uptake in aboveground biomass as a function of applied N; and the increase in yield related to the increase in N uptake (Dobermann, 2006; Jones *et al.*, forthcoming; Ladha *et al.*, 2005; López-Bellido and López-Bellido, 2001). These methods require a cultivated plot without N additions (control plot) to be compared with the fertilized plot (or a few of them receiving different application rates). There are approaches allowing precise estimation of N use and allocation by using an isotope tracer ($^{15}N$). All these approaches are useful to understand biogeochemical processes, to isolate the effect of the soil N legacy and to define optimal fertilizer rates. They require crop and soil data, investments and experimentation (Quan *et al.*, 2021), and they are applied in field trials. When working at large spatial scales, from watershed to global, or at the field scale without a field trial, the partial nutrient balance approach to NUE, which refers to the sum of N outputs divided by the sum of N inputs, is the commonly used approach (Fixen *et al.*, 2015). This indicator is complementary to N surplus, which is estimated as the sum of N inputs minus the sum of N outputs per unit of area, and it is used to analyse the agro-environmental performance of a cropping system about N use. Additional new indicators, such as Fertilizer Dependency (Quemada and Lassaletta, 2024) or circularity indicators (van Loon *et al.*, 2023), complement NUE indicators by providing information on N external dependency, recycling and circularity.

The European Nitrogen Experts Panel proposes a general framework for NUE based on the mass balance principle, which includes a graphical tool to interpret and communicate results (EUNEP, 2015). It consists of a simple two-dimensional input–output diagram where the three interpretation types are represented (Figure 2). Several lines delimit spaces, indicating system performance about multiple attributes.

FIGURE 2
**Framework of the NUE two-dimensional input–output diagram**




*Note:* Possible NUE, productivity and surplus targets, and definition of the characteristic operating space (COS). Minimum productivity and maximum surplus are defined based on expert criteria and regional knowledge and legislation. When including the 90 percent NUE threshold, the COS is delimited, and 45 percent of the farms are placed inside, indicating good agro-environmental performance.

*Source:* Authors' elaboration based on **EUNEP**. 2015. Nitrogen use efficiency (NUE) is an indicator of the utilization of nitrogen in food systems. Wageningen University, Alterra, Wageningen, the Kingdom of the Netherlands and **Quemada, M., Lassaletta, L., Jensen, L.S., Godinot, O., Brentrup, F., Buckley, C., Foray, S. *et al.*** 2020. Exploring nitrogen indicators of farm performance among farm types across several European case studies. *Agricultural Systems*, 177: 102689. https://doi.org/10.1016/j.agsy.2019.102689

A line of minimum productivity can be drawn and should be adapted regionally, locally or according to user preference. The red line represents an established maximum desirable surplus. This threshold must be established based on local vulnerabilities and/or adaptation of legislation. Fields or farms above the 90 percent NUE line are at risk of soil N mining, while fields or farms below 50 percent NUE risk serious N pollution. European Nitrogen Experts Panel (EUNEP) has established a guidance document to assess NUE at the farm level; it includes a three-tier approach (from default values to empirical figures) for several budget components, the choice of which depends on the quality of the available data (EUNEP, 2015). This document proposes a well-defined list of inputs and outputs for various systems for which the calculation can be adapted to data availability. These lines delimit a desirable space for acceptable production, low pollution risk, and without mining soil N reserves. This space has been defined as the characteristic operating space (COS).

Quemada *et al.* (2020a) illustrate an application of the two-dimensional input–output diagram and the COS approach with data from 83 farms under irrigated conditions in Spain. Most of the rotations (two or three years) include one cereal (maize) and tubers or oil crops. The surplus threshold, 50 kg of N per ha per year, is established based on expert criteria and can be modified according to new legislation or an additional environmental protection indication. They establish the minimum productivity threshold as the 75 percent quartile, but it could be modified based on changes in prices or other agronomic considerations. In conjunction with the 90 percent NUE limit, the COS is now defined. With these criteria, 45 percent of the farms had their NUE in the COS, while the rest failed for various reasons: 6 percent risked soil mining, 25 percent were not productive enough, 30 percent had a high risk of pollution, and some of them had low NUE. The question now is how to make each farm reach the COS, given their current performance. In this chapter, several strategies to improve the agro-environmental performance of cropping systems are proposed (section 2.4). For assessments at the national level, NUE features as one of seven sub-indicators of the SDG 2.4.1 proxy, used by FAO to report on the productivity and sustainability of agriculture (FAO, 2024c), in line with the COS thresholds discussed above.

## 2.3 TRENDS AND STATUS OF NITROGEN USE EFFICIENCY

### 2.3.1 Global trends

Global crop yield has been rising steadily from an average of 19 kg N/ha/yr in 1961 to 65 kg N/ha/yr in 2022 (FAO, 2023a). In contrast, NUE declined from 56 percent in 1961 to 40 percent in the 1980s and has since increased again to 56 percent in 2022 (FAO, 2023e; Ludemann *et al.*, 2024). The nutrient budget ranges vary by country (Zhang *et al.*, 2021; FAO, 2023e). Regionally, crop production in Southeast Asia increased more than threefold from 18 kg N/ha/yr in 1961 to 54 kg N/ha/yr in 2022, and in North America, it almost quadrupled, from 22 kg N/ha/yr to 80 kg N/ha/yr. As for NUE, in Southeast Asia, it decreased significantly from 65 percent in 1961 to 45 percent in the 1990s, to increase again to 54 percent in 2022. In North America, NUE first decreased from 65 percent in 1961 to below 50 percent in the 1980s, then increased to 69 percent in 2022. Nitrogen use efficiency varies at the crop level (Zhang *et al.*, 2015b). For example, soybeans had an NUE as high as 80 percent in 2010, while fruits and vegetables had NUEs as low as 14 percent in the same year. Rising productivity was accompanied by increasing synthetic N fertilizer input and N surplus, indicating greater environmental stress caused by crop production.

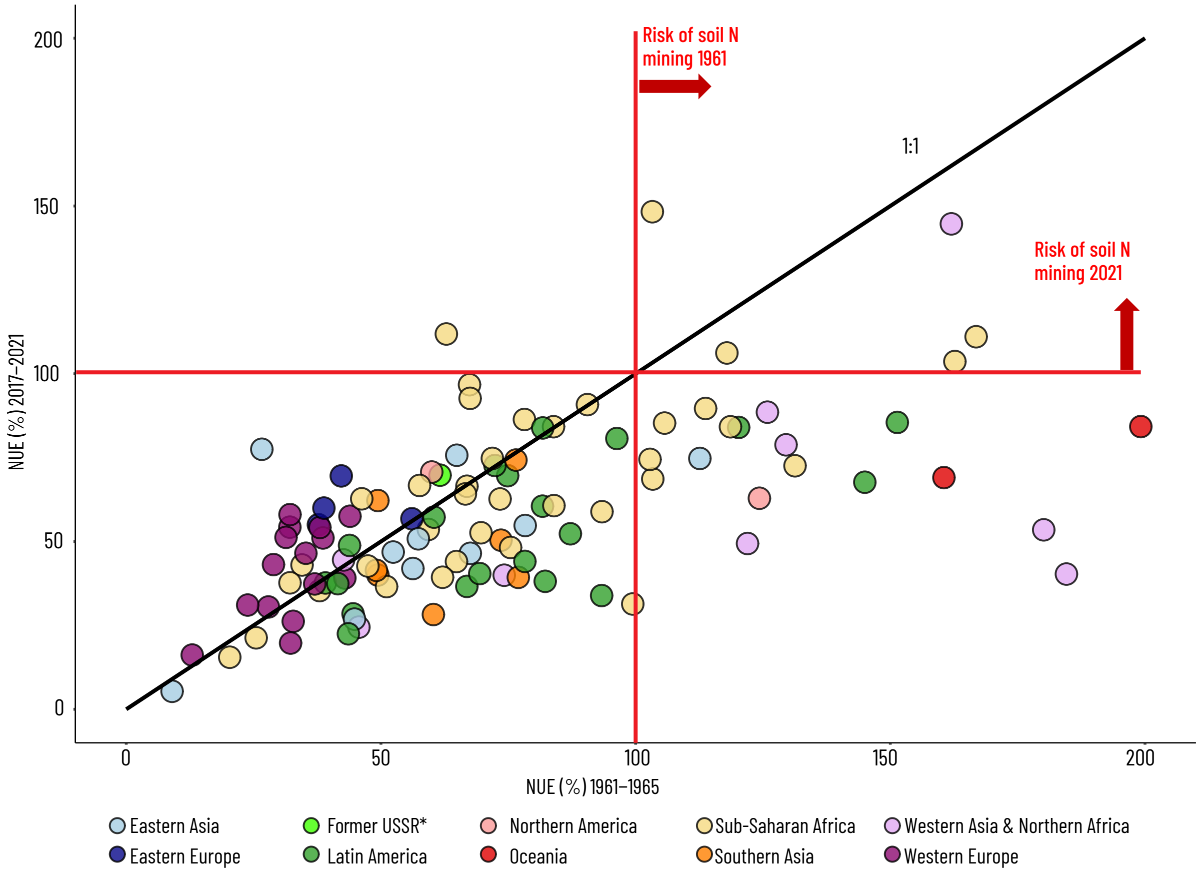
### 2.3.2 Spatial patterns

Based on the benchmarking global database of FAOSTAT (FAO, 2023e), the average global NUE (for the period 2017–2021) was 62 percent but varied substantially around the world, affected by both the ecological and socioeconomic conditions of each country (Figure 3). The patterns exhibited by countries can be categorized into five groups, as follows.

1. ***Low-yield-high-NUE.*** Many countries in sub-Saharan Africa and Western Asia had limited use of mineral N fertilizer during this period. According to FAO, average cropland N balance has remained low, about 0–40 kg N/ha/yr (FAO, 2023a). As a result, crop yield remained low, with NUE varying from 83 to 68 percent. Some countries experience a high average NUE of about 87 percent. Zhang *et al.* (2021) found that some crop-specific NUEs were higher than 100 percent, indicating the risk of depleting soil N stocks and land degradation, called soil mining.
2. ***Moderate-yield-moderate-NUE.*** Countries with N inputs of around 100 kg N/ha/yr and yielded around 50 kg N/ha/yr, with NUE around 50 percent (Figure 4). These countries are widely distributed around the world, such as in Oceania and Eastern Europe. One of the major mineral N fertilizer users in this group is the Russian Federation, which reached an NUE of around 85 percent in 2017, partly benefiting from rich soil N stocks and high N inputs and genetic improvement (Figure 3) (FAO, 2023a; Zhang *et al.*, 2021).
3. ***High-yield-moderate-NUE.*** A cluster of countries, including in North America, Western Europe and Latin America, achieved high crop yield at around 100 kg N/ha/yr with modest N inputs (Zhang *et al.*, 2021), resulting in a moderate NUE level of about 51–71 percent (FAO, 2023a). Such achievement benefited from advancements in technologies and management practices in crop production, as well as favourable climate and soil conditions for crop growth. The countries producing a quantity of soybean, which can biologically fix N, have achieved a high NUE.

FIGURE 3

**Crop nitrogen use efficiency at the country scale (1961–1965 vs 2017–2021)**




* Former USSR refers to Former Union of Soviet Socialist Republics.

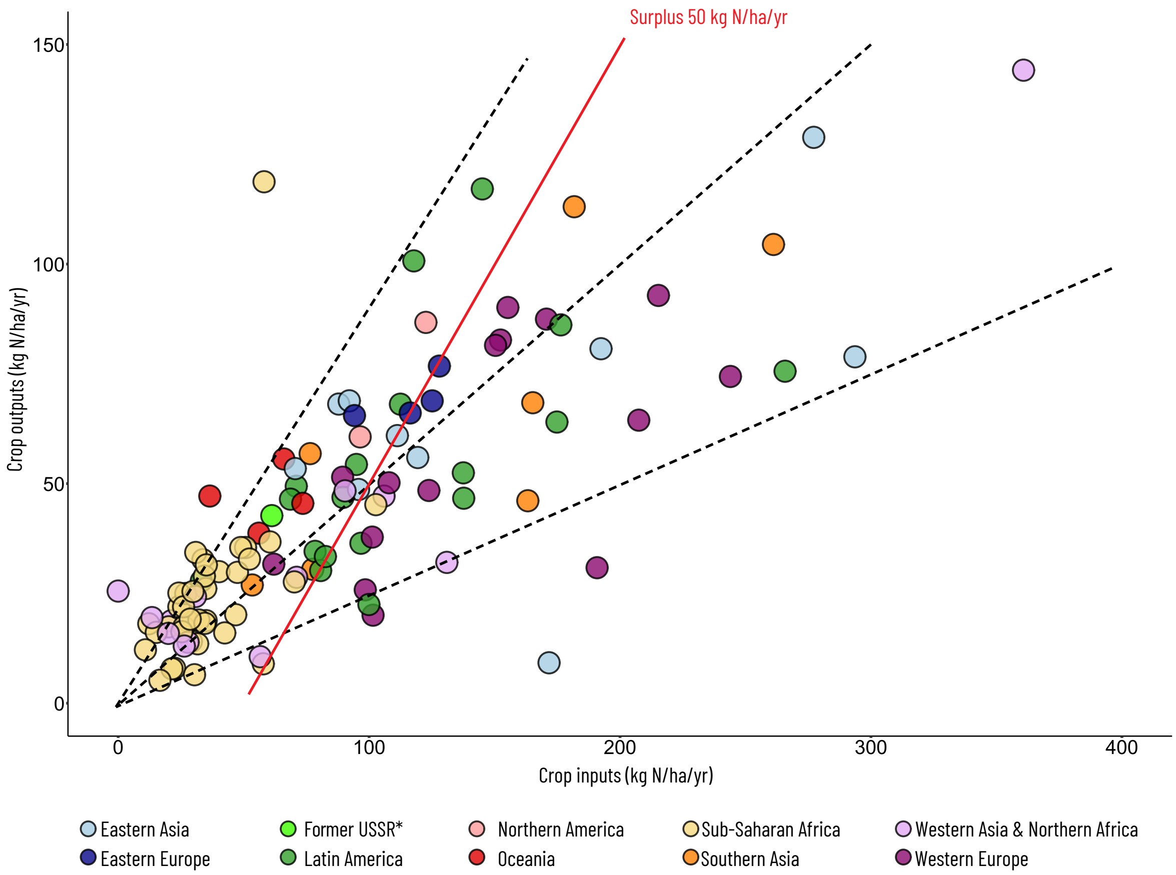
*Note:* The horizontal and vertical red lines indicate the threshold of average country crop NUE above which there is a risk of soil N mining.

*Source:* **FAO.** 2023. Crop Nutrient Balance. *FAOSTAT*. https://www.fao.org/faostat/en/#data/ESB

4. ***Moderate-yield-low-NUE.*** Compared with countries in the "High-yield-moderate-NUE" category, some countries in Southern Asia had similar N input levels, but yields were only half, resulting in NUE ranging from 37 to 63 percent (FAO, 2023a). Southern Asia is among the top mineral N fertilizer users in the world. The intensification of crop production in this region has been largely driven by intensive fertilizer inputs, but the yield response has become stagnant. Improving N management practice and reducing the incentives for excessive N application is critical for closing the yield gap and increasing NUE.

5. ***High-yield-low-NUE.*** A final group of countries in Southeast Asia and Western Europe have intensive cropland N input (e.g. above 200 kg N/ha/yr) (Zhang *et al.*, 2021). While the intensive N input has almost maximized the yield at the existing technological and ecological conditions, it resulted in low NUE and substantial N losses to the environment. Most of these countries are mid- to high-income countries but with low per capita cropland area and dense populations, indicating tremendous pressure to ensure food security with limited domestic land resources (see Figure 4).

FIGURE 4
**Nitrogen budgets in cropping systems at the regional scale (2017–2021)**




* Former USSR refers to Former Union of Soviet Socialist Republics.

*Note:* Crop inputs per ha include synthetic fertilization, manure, biological fixation and atmospheric deposition. Crop outputs correspond to harvested N. The red line represents an established maximum desirable surplus. The dashed lines indicate different thresholds for the risk of soil N mining (on the left side) to the risk of N pollution (on the right side).

*Source:* **FAO.** 2023. Cropland Nutrient Balance. *FAOSTAT*. https://www.fao.org/faostat/en/#data/ESB

### 2.3.3 Regional trends

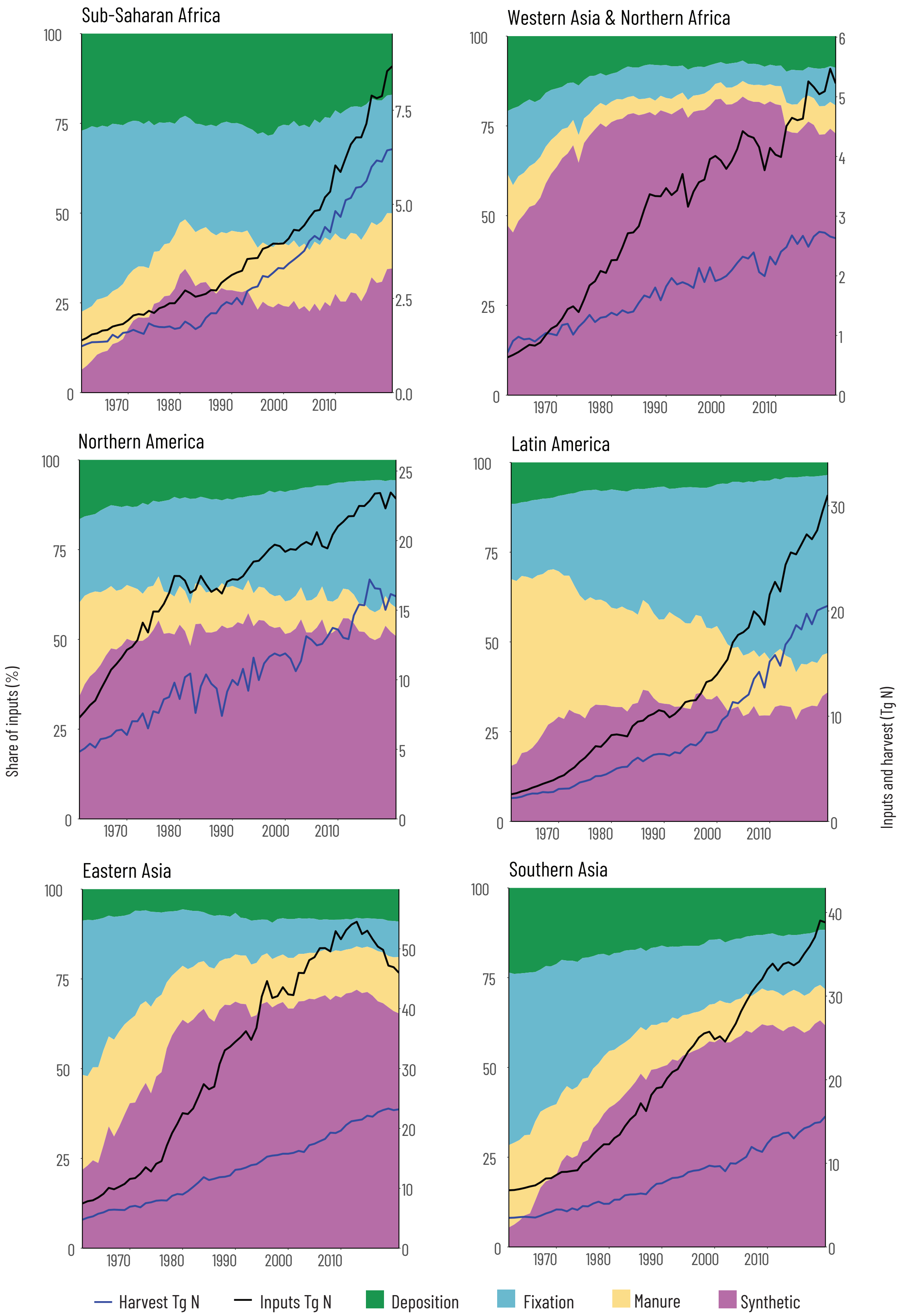
In the past decades, N inputs from synthetic N fertilizer, manure, biological N fixation, and N deposition have increased steadily, resulting in an increase in crop production in all regions of the world (FAO, 2023b; Figures 5 and 6).

Comparing NUE in the most recent five years (2017–2021) with the earliest five years (1961–1965) in the database (FAO, 2023a), most countries had lower NUE in the recent period (Figure 3). The high-income countries have higher NUE and are characterized by intensified crop production. Most countries that had a high risk of soil N mining in the early 1960s have alleviated that risk, but soil N mining remains prevalent in several countries in sub-Saharan Africa and Central Asia.

The input of N from different sources (deposition, fertilizer input, manure and N fixation) has been increasing in the last decades. For countries that have achieved a high NUE, most of them were able to improve their NUE. For instance, Northern America and Western Europe had declines in NUE during the early development stage and crop production intensification, resulting in N surplus and losses to the environment. In the 1980s, NUE started to level off and increase despite the continuing intensification of cropping (Figure 9; FAO, 2023b). The aggravating N pollution stress has slowed down in recent decades. The levelling-off of the NUE reduction in Western Europe has been driven by regulatory measures (such as the Nitrates Directive). At the same time, Northern America has mostly benefited from market incentives and voluntary measures.

For regions with lower NUE in the 2010s, most are in the early intensification stage of the U-shape trajectory (Figure 9). Some of these regions have started to level off N fertilizer use Figures 5 and 6), and consequently, the declining NUE has started to level off as well. Still, the N surplus stress has already far exceeded the planetary boundary (FAO, 2023b).

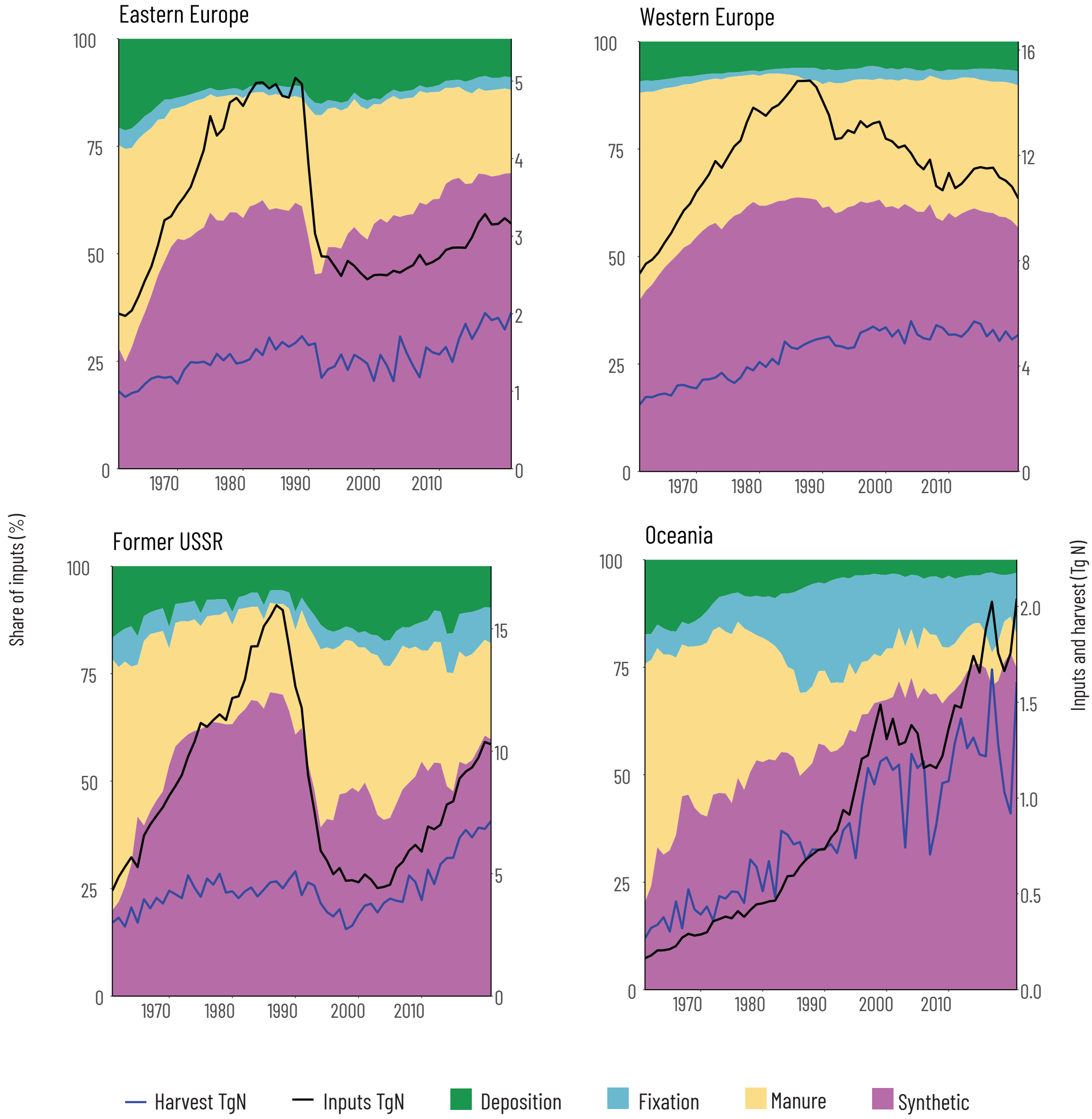
FIGURE 5
**Relative share of N inputs of major world regions**




*Note:* Relative share of nitrogen inputs is given by input category and total N inputs, and outputs by the amounts of N in harvested crops (for the period 1961–2021).

*Source:* **FAO.** 2023. Cropland Nutrient Balance. *FAOSTAT*. https://www.fao.org/faostat/en/#data/ESB

FIGURE 6
**Share of N inputs of major world regions**

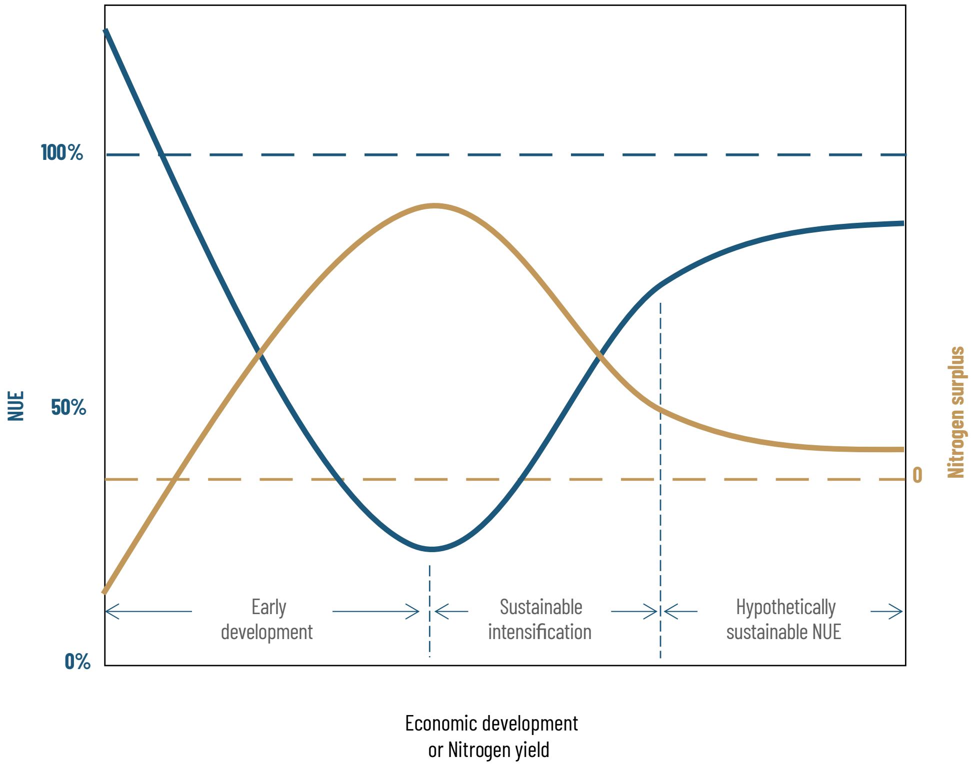



*Note:* Relative share of N inputs is given by input category and total N inputs, and outputs by the amounts of N in harvested crops (for the period 1961–2015).

*Source:* **FAO.** 2023. Cropland Nutrient Balance. *FAOSTAT*. https://www.fao.org/faostat/en/#data/ESB

The spatial and temporal patterns of NUE both show a typical U-shape pattern between NUE and yield (as well as the economic development stage). The pattern aligns with a classic Environmental Kuznets Curve theory (Figure 7), which hypothesizes that the early stage of agricultural development and crop intensification is achieved at the cost of the environment with more N inputs and lower NUE; as the economy develops further with more technological options and better environmental awareness, the economic development and crop production will rely more on efficient use of the resources such as N fertilizer and land (Zhang *et al.*, 2015b). While such relationships have been identified in most countries around the world, it should not be assumed that NUE will automatically change as the gross domestic product (GDP) grows. It is critical to learn from the lessons and experiences of those developed countries that exhibited the full course of the Environmental Kuznets Curve trajectory and identify opportunities for technological transfer and policy intervention to facilitate early intensification in countries without further sacrificing NUE and increasing N surplus.

FIGURE 7
**Environmental Kuznets Curve for nitrogen surplus and NUE**




*Note:* The graph shows an idealized environmental Kuznets Curve for N surplus and a related curve for (NUE).

*Source:* Authors' elaboration based on **Zhang, X., Mauzerall, D.L., Davidson, E.A., Kanter, D.R. & Cai, R.** 2015. The economic and environmental consequences of implementing nitrogen-efficient technologies and management practices in agriculture. *Journal of Environmental Quality*, 44(2): 312–324. https://doi.org/10.2134/jeq2014.03.0129

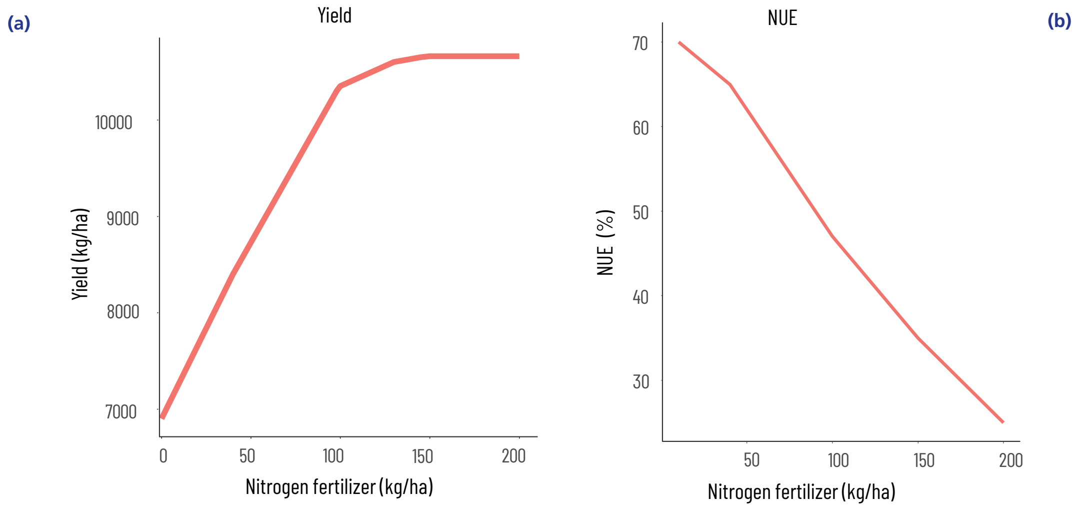
### 2.3.4 Nitrogen use efficiency and technologies

Improving technologies and management practices (TMP) has often been considered the major option for improving NUE, and tremendous efforts have been devoted to developing and identifying best management practices to balance the production and pollution-mitigation goals in crop production. It is critical to recognize that socioeconomic factors, such as market price, farmers' attitudes towards risks, fertilizer subsidies, and government price fixing, are critical for determining NUE outcome of TMP implementation (Zhang *et al.*, 2015a). It is particularly important to understand the impacts of these socioeconomic factors in the current global market with volatile mineral fertilizer and crop prices. Some examples of TMP include the use of slow-release, nano and coated urea fertilizers, precision farming, the use of bio-stimulants to enhance N crop uptake, integrated plant nutrient management, and conservation agriculture (Das *et al.*, 2021).

For a given farm and TMP level, yield response to N inputs typically levels off as N inputs increase (Figure 8a; known as the diminishing return in yield response function) because other limiting factors for yield increase become more important (e.g. water). Consequently, NUE decreases as N inputs increase (Figure 8b). The level of N application rate is determined by socioeconomic factors, such as the market price of fertilizer and crop and the farmer's risk preferences. For a given TMP, NUE is not constant; rather, it can vary considerably with fertilizer and crop prices and fertilizer application rates.

Technologies and management practices may change the yield response function in different ways, consequently impacting NUE. Taking the evolution of N management in the US corn production as an example, the yield growth during the period of 1960–2011 can be considered as three stages corresponding to NUE change:

FIGURE 8
**The role of technology in changing yield response to N inputs and NUE improvement**




*Note:* For (a), the line represents average yield response functions derived from the record of N fertilizer yield, and (b), represents the average NUE in function of the N fertilizer inputs for corn in the United States.

*Source:* Authors' elaboration based on **Zhang, X., Davidson, E.A., Mauzerall, D.L., Searchinger, T.D., Dumas, P. & Shen, Y.** 2015. Managing nitrogen for sustainable development. *Nature*, 528(7580): 51–59. https://doi.org/10.1038/nature15743

- From 1961 to 1971, yield increase almost followed similar yield production functions, suggesting that yield increase was mainly achieved by adding more N, and NUE decreased.
- From 1971 to 2001, yield increase was mainly achieved by adopting technologies that require more N input, and NUE levelled off or decreased.
- From 2001 to 2011, yield increase was mainly achieved by adopting technologies that require similar or less N, and NUE increased.

Consequently, identifying and implementing TMP that increase or maintain yield without requiring additional N inputs are critical for ensuring a positive outcome for NUE improvement.

### 2.3.5 Nitrogen use efficiency by crop type

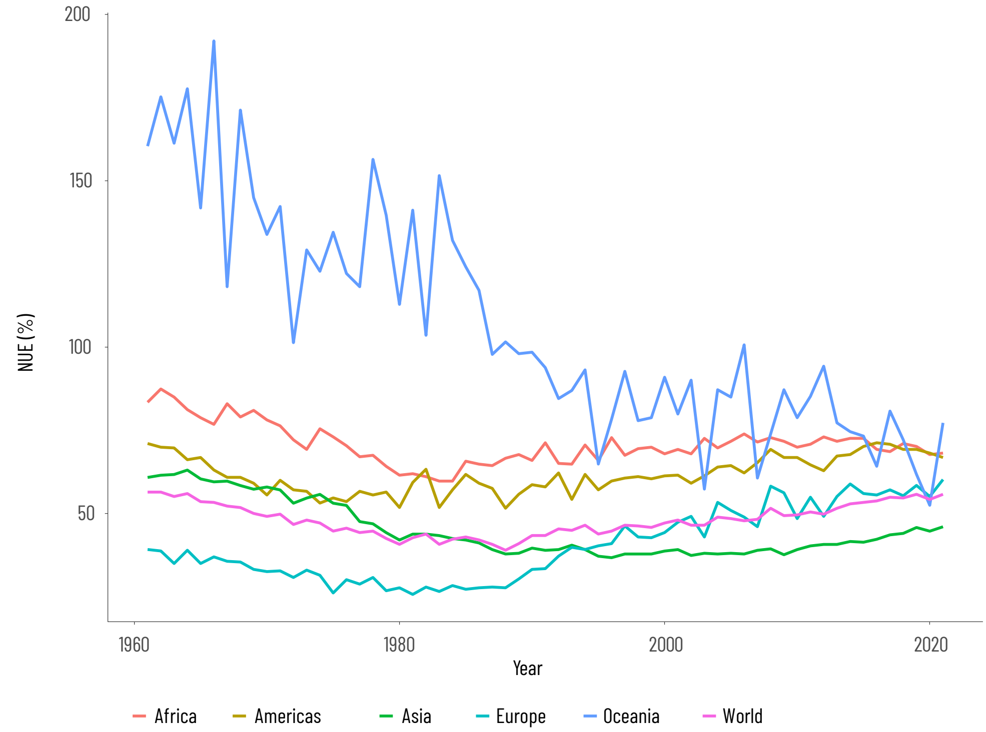
The crop NUE at the country level is affected by the performance of a mixture of crops that are produced in each country, in addition to the general trends of economic development and associated technological advancement. Even for the same crop, NUE can vary greatly between regions and practices. It is recognized that most leguminous crops, such as soybean and alfalfa, typically have higher NUE than other staple crops because they can biologically fix N and rely on a few new N inputs. Fruits, vegetables and sugar crops typically have NUE below 20 percent, lower than major staple crops. Zhang *et al.* (2015a) demonstrated that even if countries in Southeast Asia increased NUE for each crop type to the Northern American level, their overall NUE would be much lower than that in Northern America. The reason is that Southeast Asia has devoted about one-third of its fertilizer to low-NUE fruit and vegetable production, while Northern America has produced soybean as one of its major crop products. With the call for increasing access to affordable, healthy diets, reducing the consumption of energy-dense and highly processed foods, and maintaining a balance between food groups, including an abundance and variety of fruits and vegetables, it is imperative to improve NUE for these crops.

Data for NUE by country and crop type is still scarce and highly uncertain. FAO has disseminated (in the FAOSTAT database) a time-series estimate of NUE per country and region (Figure 9), (FAO, 2023b). Future efforts are needed to improve data availability and quality to better evaluate NUE for each crop type, environmental trade-offs, international trade patterns and production performance (Mueller *et al.*, 2017; Yao *et al.*, 2021; Zhang, 2017).

## 2.4 IMPROVEMENT OPTIONS TO ENHANCE NITROGEN USE EFFICIENCY

Optimizing NUE requires the combination of improved N fertilizer use and management and the implementation of other practices that contribute to improving the crop status and the potential to achieve higher productivity, which is closely linked to enhanced recovery of nutrients in the "crop pool" (above- and below-ground crop biomass rather than other pools or N loss pathways). Apart from the substantial policy and regulatory adjustments required to address N challenges (see Chapter 6), several technologies

FIGURE 9
**Trends of NUE by regions and the world (1961–2021)**




*Note:* The graph shows estimated NUE in cropping systems from 1961 to 2021 for all regions of the world and an aggregated estimate of NUE for the world. The values of NUE above 100 percent indicate the mining of soil nitrogen.

*Source:* Authors' elaboration based on **FAO.** 2023. Cropland Nutrient Balance. *FAOSTAT*. https://www.fao.org/faostat/en/#data/ESB. Accessed 22 September 2024.

and practices can increase NUE at the farm level and are described in section 2.4.1 (see 2.5.2 for a case study describing different technologies and practices for crop production in Mexico). The improved use of fertilizers, usually known as the 4Rs approach (which means applying N fertilizers with the right rate, right placement, right time and right source; Bruulsema, 2018), is a set of valuable and comprehensive principles described in more detail in section 2.4.2. Finally, section 2.4.3 focuses on nature-based solutions to increase NUE in cropping systems.

### 2.4.1 Management practices to increase nitrogen use efficiency at the cropping scale

Improving NUE should be focused on practices that display synergies with N fertilization, which is a key management practice to enhance NUE (You *et al.*, 2023). These synergistic practices can be classified as follows.

1. **Improvement of crop N status and nutrient acquisition potential:** These practices aim to decrease the amount of reactive N in the soil (liable to be lost to the environment) while enhancing the recovery of N (and other nutrients) in crop biomass. These strategies include crop mixtures or intercropping with N-acquisitive species (Abalos, van Groenigen and de Deyn, 2018), crop breeding conducted to increase N uptake, management of crop density, balanced fertilization or integrated nutrient management plan (with a sufficient supply of macronutrients, such as phosphorus [P] and potassium [K], and micronutrients, with special attention paid to those with synergies with N acquisition) and the use of biostimulants and/or biofertilizers (including plant growth promoting microorganisms) (Ferreira, Soares and Soares, 2019; Sutton *et al.*, 2022).
2. **Natural alteration of N cycling:** including the use of green manures (including fertilizer trees such as acacia, *Sesbania sesban*, *Gliridicia sepium*) (Sileshi *et al.*, 2014), organic mulch, N-fixing microbes and biological nitrification (and denitrification) inhibition (Galindo *et al.*, 2021; Galland *et al.*, 2019; Saud, Wang and Fahad, 2022). Soil microbial communities play a key role in the regulation of other N cycle processes, such as mineralization/immobilization or the stabilization of

organic N in the soil (Beed *et al.*, 2011). The use of organic fertilizers, slow-release and coated fertilizers are considered more environmentally benign than conventional fertilizers (Pan *et al.*, 2016). Current technologies and practices include stimulation of the dissimilatory $NO_3^-$ reduction to ammonium ($NH_4^+$) (Dimkpa *et al.*, 2020) and nano fertilizers (Zhou *et al.*, 2017), which are effective in improving NUE. The effectiveness of these strategies largely depends on soil conditions, agriculture management, and climate (Dimkpa *et al.*, 2020). The key often lies in customizing N management solutions and technology to the specifics of cropping systems and the starting N situation in a specific region.

3. **Improvement of soil health and fertility:** involving conservation agriculture (e.g. reduced or zero tillage, crop rotation, cover crops) or the application of organic fertilizers, compost and biochar (effective in the mitigation of $NO_x$ and N leaching). These practices can mitigate N losses (Liu *et al.*, 2018) and/or reduce the dependence on external inputs such as synthetic fertilizers (Kaye and Quemada, 2017). The crop use efficiency of exogenous nutrients (including N) strongly depends on soil health (such as the ability of the soil to sustain the productivity, diversity and environmental services of terrestrial ecosystems). The technical and economic investment linked to organic or synthetic fertilization is threatened if the soils lack proper physical (erosion, compaction), chemical (salinity, hydrogen ion concentration [pH], nutrient or SOM depletion), or biological (contamination, loss of biodiversity) quality. Soil health should be a primary focus in sustainable N management. Furthermore, soil fertility and N monitoring in N management can improve decision-making.
4. **Irrigation management or modification of evaporation rates:** for example, the use of mulching to reduce $NH_3$ losses (Sha *et al.*, 2021), irrigation dose adjusted to crop needs (thus decreasing potential drainage water), deficit irrigation and implementation of irrigation systems that improve water use efficiency (e.g. micro-irrigation, subsurface irrigation) while reducing reactive N losses (Kuang *et al.*, 2021; Quemada *et al.*, 2013).

### 2.4.2 Fertilization management to improve nitrogen use efficiency at the cropping scale

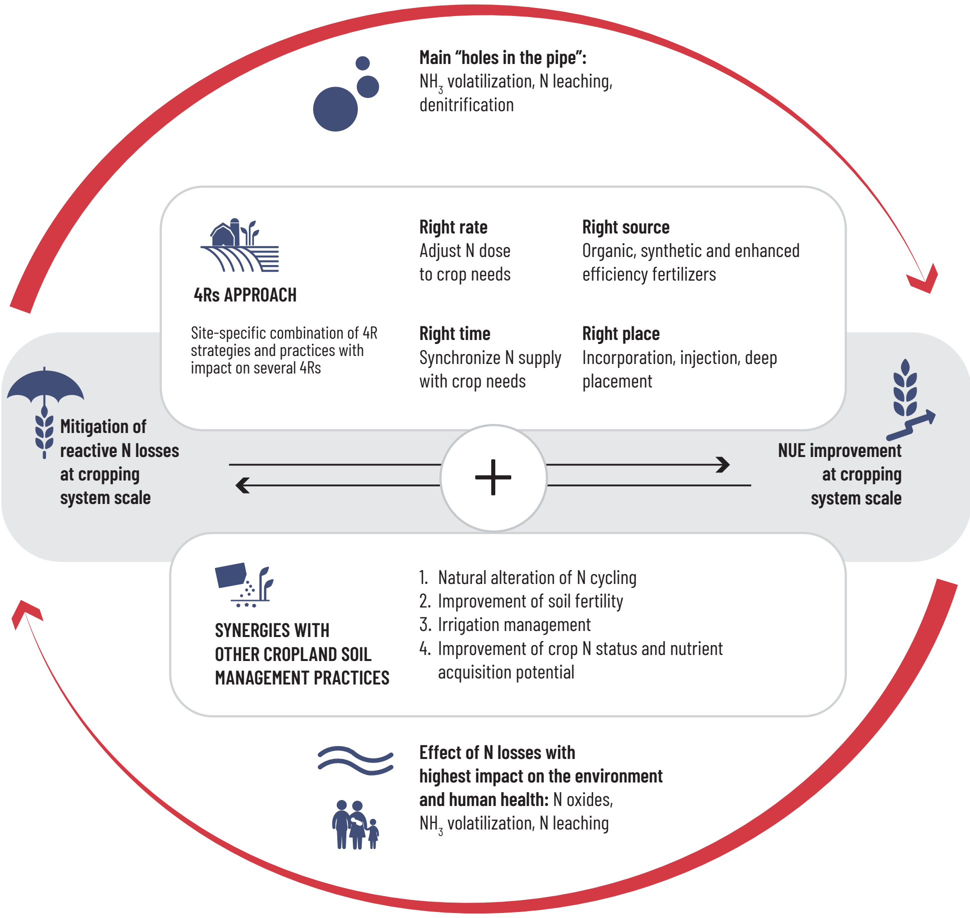
The improved use of fertilizers, usually known as the 4Rs approach, is highly effective in increasing NUE (Bruulsema, 2018; Fixen, 2020). These practices are centred on the mitigation of N losses with the highest quantitative relevance (depending on environmental or management conditions): $NO_3^-$ leaching, $NH_3$ volatilization and N losses through denitrification (Sutton *et al.*, 2022, 2013). The 4Rs approach focuses on the right rate, the right placement, the right source and the right timing of fertilization (Figure 10).

The **right rate**, that is the adjustment of the N dose to crop needs, decreases gaseous, leaching and runoff losses globally, with neutral or negative effects on crop yields and abatements of N uptake, ultimately increasing NUE (van Grinsven *et al.*, 2022; Xia *et al.*, 2017). In several cropping systems (e.g. cereal crops) and areas, the main N source for plants can be organic or synthetic fertilizers, as well as the turnover of soil and crop residue N (Yan *et al.*, 2020). Here, comprehensive integrated soil fertility management can help adjust or reduce synthetic N fertilizer needs. This would result in economic and environmental benefits such as the reduction of upstream GHG emissions from fertilizer manufacturing (a major component of the carbon footprint of agrifood products).

The **right placement** of N has a critical influence on N losses via $NH_3$ volatilization (which, on average, accounts for nearly 18 percent of the applied N but with substantial variability) (Pan *et al.*, 2016). The incorporation (i.e. mechanical or through irrigation) of urea, slurries or solid manure is one of the most promising strategies for increasing NUE while decreasing yield-scaled $NH_3$ volatilization and surplus (i.e. the difference between N input and N in the harvested parts of the crop, which is an indicator of potential pollution). At a global scale, abatements of reactive N losses have been described for band spreading, trailing shoe application, injection, irrigation, mechanical incorporation and particularly for deep placement (Sutton *et al.*, 2022; Xia *et al.*, 2017).

The use of the **right N source** involves several decisions, such as the comparison between organic and synthetic sources, the comparison between ureic, ammonium-N- or nitrate-N-based fertilizers and the use of enhanced-efficiency fertilizers (such as slow/controlled release of fertilizers, nitrification and/or urease inhibitors) (Li *et al.*, 2021; Thapa *et al.*, 2016). Enhanced-efficiency fertilizers are valuable technologies to reduce the loss of nutrients and provide higher NUE. The development of biodegradable coatings (such as agricultural residues, biochar, starch, lignin, chitosan and alginate) and natural inhibitors (e.g. neem-coated urea) is currently of increasing interest to improve the adoption/applicability of these compounds, minimize the negative impacts on soil biological quality and reduce the use of microplastics or synthetic molecules of uncertain degradability (Jariwala *et al.*, 2022; Singh, 2016). Critical concerns about the use of organic fertilizers are the availability of N, and the synchronization between N supply and crop needs to avoid yield and NUE penalties. A combination of organic and synthetic fertilizers could help minimize negative side effects on yields, optimize NUE, and avoid pollution swapping derived from the excess of other elements such as phosphorus (Liu *et al.*, 2021). The choice of the right N source should be

FIGURE 10
**Overview of the 4Rs nutrient stewardship paradigm**




*Note:* The 4Rs nutrient stewardship aims to minimize N losses and improve NUE. The potential synergies of 4Rs with other agronomic practices and the relationships between NUE and the mitigation of reactive N losses are shown.

*Source:* Authors' elaboration.

site-specific and matched to management, edaphic and climatic conditions. For instance, in conditions with a high risk of leaching and denitrification (e.g. humid grasslands, coarse-textured soils with good drainage, and crops with a high water demand), nitrate-based fertilizers should be replaced with controlled-release technologies and/or ammonium-N-containing or ureic fertilizers (Xia *et al.*, 2017). In basic-pH soils with a high risk of volatilization and nitrification losses, nitrate-based fertilizers, or those containing urease and/or nitrification inhibitors, have been shown to reduce pollutant N losses, compared with urea alone (Guardia *et al.*, 2021). There are food safety issues associated with the use of environmental inhibitors that need to be considered and addressed, as elaborated by FAO (2023c). See section 2.5.1 for a case study on nitrification and urease inhibitors.

Applying N at the **right time** aims to synchronize N availability and crop demand, decreasing the chances of losing reactive N to the atmosphere or groundwater.

Sustainable improvements of NUE require the site-specific combination of several 4Rs strategies or the use of practices that involve the application of more than one "R" (Liang *et al.*, 2019; Nasielski *et al.*, 2020). Examples of these practices are the use of controlled-release technologies or fertilizers with inhibitors (which are the "right" sources to decrease some reactive N losses as well as improve synchronization – right time – with crop N demand), fertigation (which helps to improve the right placement and right time) or precision agriculture (which acts mainly on the right rate and placement). Among precision agriculture techniques, the use of variable-rate N fertilization can optimize crop N status and decrease N surpluses, while potentially increasing energy use efficiency and farm profitability (Jovarauskas *et al.*, 2021; Pedersen *et al.*, 2021). Variable-rate fertilization can be based on real-time or near-real-time monitoring combined with top-dressing fertilization (e.g. tractor-based sensors, manual sensors/smartphones-based systems on remote sensing information) and/or on static information derived from soil data, remote sensing (including satellite-based sensing) or yield maps from previous years (Bacenetti *et al.*, 2020). Nitrogen use efficiency related to precision agriculture has the challenge of accurate and timely detection of plant N requirement and supplying fertilization accordingly, which still requires closing the gap between spectral information and biomass status and improving the distinction between N deficiency and other abiotic stresses. The extension of soil analyses and advisory facilities is a valuable tool for combining 4Rs practices and optimizing NUE.

In addition to the effects of 4Rs strategies on NUE and crop yields, it is essential to consider the side-effects on GHG balance (including carbon [C] sink in soils), crop yield and quality, and the net economic benefit, as well as the potential barriers (social, technical, economic, etc.) and opportunities for the adoption of 4Rs-based technologies and management practices (Sanz-Cobena *et al.*, 2017). Cropping surface, the lack of agronomic capacity to address nutrient losses, land tenure, and policy (e.g. crop insurance), and biophysical factors (e.g. slope) are key factors driving or constraining the widespread adoption of 4Rs (Upadhaya, Arbuckle and Schulte, 2023).

### 2.4.3 Nature-based solutions to optimize nitrogen use in cropping systems

Nature-based solutions can be viable and cost-effective options for pursuing sustainable development, consistent with conserving biodiversity and natural resources. In cropping systems, this approach has the added value of supporting the maintenance of ecosystem functions and services (Arnés and Santiváñez, 2021). Healthy soils, pulses, soil microbes and agrobiodiversity are nature-based solutions that help optimize N use and reduce N imbalances, and are beneficial for nutrient cycling and soil biodiversity. Crop nutrient replenishment can be supported through strategies (or a combination of strategies), including N-fixing leguminous crops (FAO, 2016), biofertilizers (Ibáñez *et al.*, 2023; Tosi *et al.*, 2020), recycled nutrient sources and the addition of organic matter (Geissdoerfer *et al.*, 2020; Valve, Ekholm and Luostarinen, 2020). A holistic approach to N fertilization, which includes the addition of organic matter, helps to optimize soil physical and biological properties for better assimilation and retention of soil nutrients, including N. Organic matter addition improves nutrient availability, gaseous exchange, water retention and infiltration, and is essential for the growth and development of soil biodiversity (Lal *et al.*, 2018; Lorenz and Lal, 2018; Stockmann *et al.*, 2013; Wiesmeier *et al.*, 2019). Leguminous crops and pulses can increase soil N through establishing symbiotic relationships with soil bacteria. Lentils, for example, exhibit a N-fixing capacity of 35–100 kg N/ha, which may reduce reliance on synthetic fertilizers and mitigate $N_2O$ emissions (FAO, 2016). The effectiveness of biological N fixation by legumes depends on environmental and agronomic factors, including soil conditions, with enhanced crop yields observed when combined with moderate synthetic fertilization (Abdullahi, Aliyu and Gabasawa, 2020; Giambalvo *et al.*, 2004; Köpke and Nemecek, 2010; Peoples, Boddey and Herridge, 2002). Biofertilizers constitute another way to benefit from nature. For instance, the use of biofertilizers containing arbuscular mycorrhizal fungi and P-solubilizing and N-fixing microorganisms has shown promise in increasing agronomic yields and NUE (Schütz *et al.*, 2018). However, some commercial microbial inoculants have been less effective in enhancing root colonization and crop growth (Koziol, Lubin and Bever, 2024). The efficacy of biofertilizers is influenced by climatic conditions and SOM availability, with reported optimal performance in arid climates (Tosi *et al.*, 2020). Weather alerts, along with the above strategies, can be powerful tools for farmers and technicians in increasing NUE, provided they are accessible, accurate and timely, easy to understand, and locally adapted (Agyekum, Antwi-Agyei and Dougill, 2022; Nepal *et al.*, 2024).

Farming practices to increase NUE should include the reduction of $NO_3^-$ leaching from croplands to minimize degradation of ground and surface waters. This can be effectively ensured through practices that closely monitor and manage soil water and N status over the cropping season. In this context, drip irrigation and fertigation technologies are valuable since they increase both irrigation-water and NUE efficiency, which in turn decrease $NO_3^-$ leaching. Additional practices to reduce $NO_3^-$ migration to water bodies are cover crops, which can scavenge residual N from soil; conservation tillage, which enhances water infiltration and reduces surface runoff; and the establishment of buffer strips and riparian zones along water bodies to capture and filter excess nutrients. Mapping the leaching potential of regions and countries as affected by soil physical properties, irrigation practice and crop management practices is needed to identify high-risk areas and allow targeted mitigation efforts. It is important to recognize that leaching plays a critical role in mitigating salt accumulation, particularly in arid climates. Extending the retention of $NO_3^-$ in the soil can provide opportunities for the denitrifying community to convert it to NO, $N_2O$ or $N_2$ depending on soil conditions.

The examples discussed above highlight how efforts to reduce one type of pollutant can unintentionally lead to increases in another. This underscores the challenge of balancing different environmental impacts and trade-offs associated with N management practices. Decision-makers need to carefully consider how interventions targeted at reducing specific forms of pollution might affect other aspects of the N cycle and overall environmental quality. Sustainable approaches to N management require holistic strategies that account for the interconnectedness of various environmental processes and aim to minimize overall pollution impacts across different environmental compartments.

### 2.4.4 Models and technology-oriented solutions to support multiple levels of nitrogen-related decision-making

Various technological solutions are used to facilitate N assessment and support decision-making at different scales. For instance, sensors that gauge crop vigour by measuring the Normalized Difference Vegetation Index (NDVI) using infrared detectors have proven valuable at the farm level of decision-making as they facilitate precision agriculture practices (Raun and Schepers, 2008). By mapping NDVI values across a field, farmers can identify specific areas that require targeted interventions, such as adjusting irrigation, applying fertilizers, optimizing resource allocation, and maximizing crop yields while minimizing inputs. In Africa, where blanket fertilizer application is often practised, decision-support tools for fertilizer recommendation are valuable to address the waste of N resources and facilitate crop growth by integrating data on soil types, nutrient levels, climate, and specific crop requirements to provide tailored fertilizer recommendations. At the global level, models such as the Nitrogen Index (Delgado and Follett, 2011) and the DayCent Century model (Parton *et al.*, 2015) are successfully used in different global regions to assess N management practices. For instance, DayCent is designed to simulate C, N and water dynamics in agroecosystems, forests, grasslands and other terrestrial ecosystems over daily to century-long time scales. Other models such as the Global Biosphere Management Model and the climate model IMAGE have been used to perform N assessments and quantify total N budgets at global and continental scales (de Vries *et al.*, 2011). While these models operate at global scales, simulations can inform broader agricultural strategies and policies that may indirectly influence farm-level practices.

### 2.4.5 Nitrogen use efficiency at the crop rotation scale

Agroecological strategies of crop diversification such as agroforestry, intercropping, cover cropping, cultivar mixture and rotations have substantial and variable benefits on biodiversity, yield improvement and ecosystem services (Beillouin *et al.*, 2021; Gaudin *et al.*, 2015a, 2015b). Crop rotations and cover crops can play an important role in the N cycle, boosting the reuse of available sources, reducing N surpluses and pollution, reducing the demand for new inputs and, in summary, increasing NUE. Specifically, diverse rotations (including cover crops) can increase cumulative N and water uptake, increase organic C inputs, water infiltration and retention while reducing $NO_3^-$ leaching (Renwick *et al.*, 2019).

While N management is scheduled at the one-year single-crop scale, the consideration of the full rotation is firmly recommended as is the definition of well-planned multiyear rotations (Lassaletta *et al.*, forthcoming). Even for monocropping systems, the evaluation of optimal fertilization rates should be estimated based on a long-term assessment of the system (van Grinsven *et al.*, 2022). This is associated with the relevant legacy effect of N from one season to the next, which occurs in monocropping systems (Quemada *et al.*, 2019; Vonk *et al.*, 2022; Yan *et al.*, 2020).

Both conventional and organic systems can include crop rotations and cover crops. In general, crop rotations of conventional systems are shorter than those in organic systems (Barbieri, Pellerin and Nesme, 2017). In conventional systems, N extracted is restored by the application of new synthetic N fertilizers, and possibly include leguminous crops, while in organic systems much of the N fertility comes from the direct incorporation of N fixed by a leguminous crop or through livestock manure. The N embedded in this manure can originate from the leguminous part of the rotation if crop and livestock systems are connected (Garnier *et al.*, 2016). Crop rotations producing fodder crops and including livestock make it possible to take advantage of local N, boosting circularity in agricultural systems. Soil health, quality and SOM content should be taken into account when considering the use of either organic or synthetic N fertilizer (Birkhofer *et al.*, 2008; Pahalvi *et al.*, 2021; Tripathi *et al.*, 2020).

Conventional rotations usually cover two or three years including cereals and in some cases leguminous crops such as the two-year soy–maize rotation of the Corn Belt in North America (Farmaha *et al.*, 2016) and Southeast Asia (Liu *et al.*, 2013), two-year wheat–maize in Southeast Asia (Liu *et al.*, 2011), two-year bean–maize in Eastern Africa (Franke *et al.*, 2018) and three years of cereals (wheat or barley) with grain and leguminous crops (e.g. peas or beans) and oilseeds (e.g. sunflower or rapeseed) in Western Europe (Anglade, Billen and Garnier, 2017; Benoit *et al.*, 2015; López-Bellido and López-Bellido, 2001; Nemecek *et al.*, 2008). Three- to ten-year rotations are established in Western European organic systems and can improve NUE substantially. Organic rotations commonly include two- or three-year rotations of N-fixing fodder crops (e.g. alfalfa or clover, temporary grasslands) introducing new N into the system. Cereals, tubers and grain-legume crops (lentils, beans, chickpeas or peas) are planned and can be combined with flax or hemp, which helps to integrate a spring crop into the rotation with low N requirements (Billen, Le Noë and Garnier, 2018; Petersen *et al.*, 2006).

Thus, when analysing the agro-environmental performance of the system including efficient N management, the full crop rotation should be considered. An isolated one-year analysis will result in misleading conclusions concerning NUE production and pollution. A high N surplus is frequently observed after grain–legume crop cultivation (Anglade *et al.*, 2015; Beillouin *et al.*, 2021). When leguminous crops are introduced, the reduced need for synthetic fertilizer for the whole rotation must be considered (Nemecek *et al.*, 2008).

Figure 11 illustrates an input–output chart including each crop of the rotation individually and considering the full rotation (Lassaletta *et al.*, forthcoming). Nitrogen use efficiency limits at 90 percent and 50 percent are shown, as well as an indicative line of maximum surplus set at 85 kg N/ha/yr (this threshold should be adapted to local conditions and vulnerabilities). Part of the high surplus estimated for some individual crops is likely not emitted to the environment but can be a valuable source of N transferred to the next crop – this must be considered in fertilization planning. Individual crops presenting high NUE (over 90 percent), which could indicate soil N mining and could be the diagnosis resulting from a one-year-only analysis, is the result of this fertility transfer.

Rotations including leguminous crops should be carefully planned, promoting practices that alleviate $N_2O$ emissions and $NO_3^-$ leaching that could be triggered after its cultivation. These practices include the mixture of cover crops (legume–cereal) (Hansen *et al.*, 2019). Crop diversification through the cultivation of cover crops replacing fallow reduces $NO_3^-$ leaching, increases soil organic C and introduces new N when there are N-fixing crops (Constantin *et al.*, 2012; Guardia *et al.*, 2019; Quemada *et al.*, 2020b). When they are cultivated and harvested, they are to be included in the N budget and NUE estimation of the full rotation.

In conclusion, the diversification of crop rotations and inclusion of cover crops is recommendable for their multiple benefits, including better N use. The rotations must be carefully planned and adapted to local agro-environmental conditions to maximize the benefits while reducing N losses. It is strongly recommended that N budgets and NUE estimations in cropping systems are based on information including the full crop rotation.

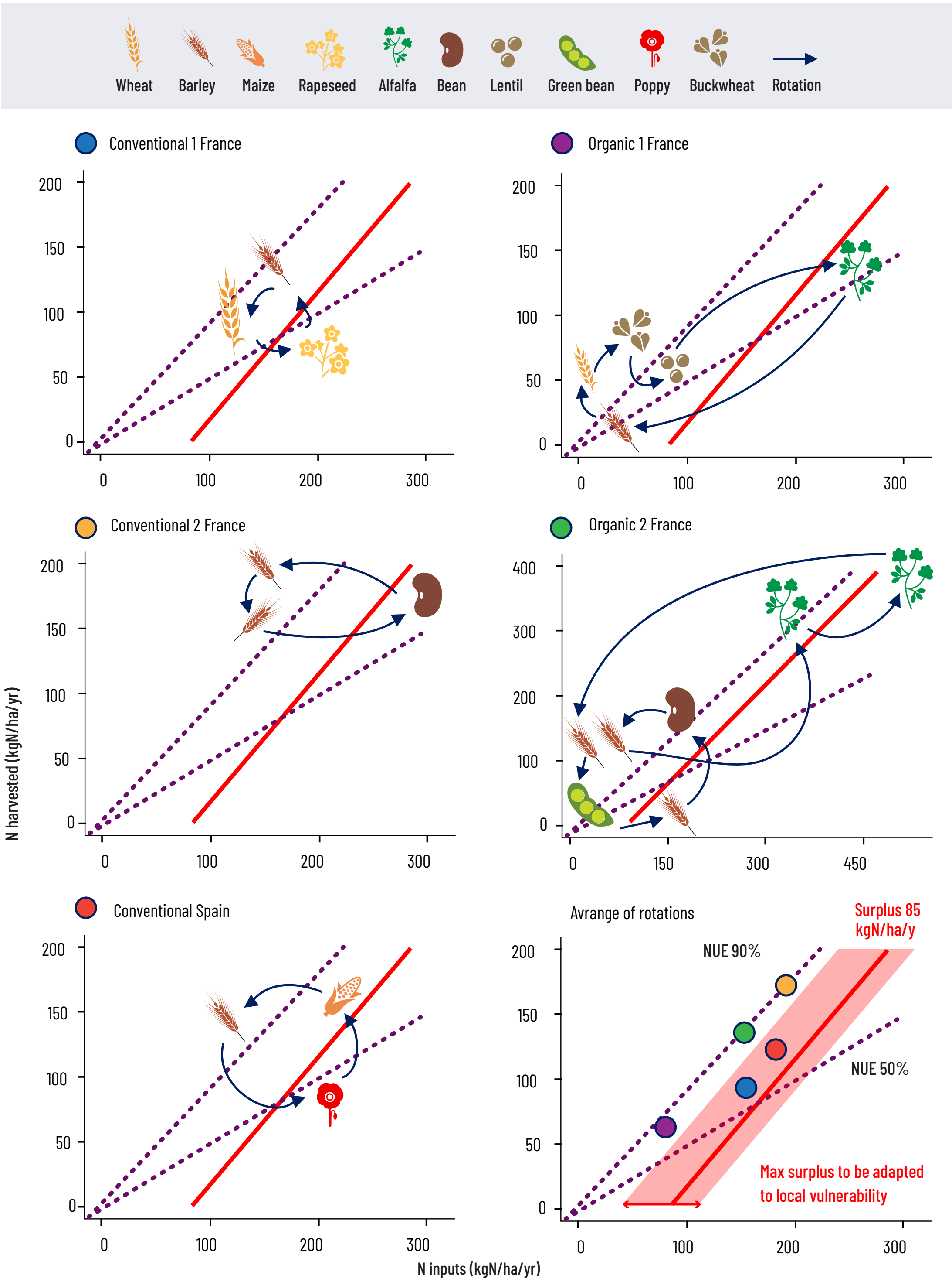
### 2.4.6 Landscape approaches to improving cropland nitrogen use efficiency

Agricultural systems are often highly specialized and regionalized, which can exacerbate N losses and inefficiencies. For example, some areas with good access to marine ports are specialized in livestock production and other separate areas are specialized in crop production (Billen *et al.*, 2010; Rodríguez *et al.*, 2023; Wei *et al.*, 2023). When crops and livestock are spatially disconnected, the agro-environmental problems associated with N use can be magnified (Bai *et al.*, 2022). On the crop side, the lack of manure availability results in a constant need for new N as synthetic fertilizer; the system becomes linear, and the applied N embedded in the grain never returns to the territory. On the livestock side, external dependency grows, and large amounts of manure cannot be efficiently reused as fertilizer in the surrounding cropping systems (Lassaletta *et al.*, 2024; Le Noë *et al.*, 2016). Thus, the reconnection of crop and livestock systems has great potential to circularize N flows in agricultural systems, simultaneously reducing the demand for new N inputs and pollution while increasing regional self-sufficiency (Simon *et al.*, 2024; Gu, 2022; Schut *et al.*, 2021). As a result, the NUE of the system will rise (Bittman *et al.*, 2023; Garnier *et al.*, 2023; Spiegal *et al.*, 2020). This topic is further discussed in Chapter 3.

Aligned with crop–livestock reconnection, and the goal of avoiding hotspots of N overuse, is the more general strategy of optimal spatial allocation of N inputs across croplands. Here the term "optimal allocation" refers to the distribution of a given amount of N inputs across a cropland system in a way that maximizes N yield and minimizes N surplus (Bodirsky and Müller, 2014; Mueller *et al.*, 2017; Zhang, 2017). If N-yield response functions are known, "optimal" N rates (from the perspective of minimizing pollution) can be calculated for the spatial unit of interest. At the field scale, optimal allocation is consistent with using the 4Rs principle of the right rate, which may vary across management zones within a field. At the farm scale, optimal allocation involves preferentially applying N according to the productivity of each field and crop. At the global scale, increasing N inputs in regions with large yield gaps and low N inputs while simultaneously decreasing N inputs in high-input regions could either increase total N production, decrease total N pollution, or achieve some combination of these objectives (Mueller *et al.*, 2017). Most farmers optimize for profit, not NUE, and coordinated shifts in N management practices are difficult in most cases; nevertheless, this approach can help identify opportunities across scales for more efficient N use (Zhang, 2017). Optimal N fertilizer application rates should consider other limits on crop productivity and the need for balanced fertilization with other nutrients, including P and K (Ren *et al.*, 2022). Balanced fertilization requires considering the full nutrient composition of manure resources (Bouwman *et al.*, 2017).

The reuse of the N embedded in human excreta as fertilizer has the potential to recover part of the N embedded in human food (Billen *et al.*, 2021). Recent studies highlight that available technologies could recover 70–80 percent of N in human excreta, particularly in the urine where N is concentrated (Martin *et al.*, 2022; Patel, Mungray and Mungray, 2020). Assuming a 70 percent recovery rate, if well treated, 23 Tg N could be recovered as fertilizer at the global scale, representing more than 20 percent of synthetic fertilizer use. Potential replacement of synthetic fertilizers would contribute to increasing NUE of the agrifood system while decreasing GHG emissions associated with the production of synthetic fertilizers, which are estimated to account

FIGURE 11
**Comparison of N inputs and outputs in harvested crops in five cropping systems under conventional and organic management in France and Spain**




*Note:* Each point represents one year of cultivation. Blue arrows show the temporal direction of the rotation. The red line represents 85 kg N/ha/yr of surplus. Dotted lines are NUE limits at 90 percent and 50 percent. This amount can be excessive in areas with low percolation or in steady-state soils without net N sequestration in the soil organic matter. The lower-right panel shows the aggregated input–output figure for the whole rotation.

*Source:* Authors' elaboration based on **Anglade, J., Billen, G. & Garnier, J.** 2017. Reconquérir la qualité de l'eau en régions de grandes cultures: agriculture biologique et reconnexion avec l'élevage. *Fourrages*, 231: 257–268. https://afpf-asso.fr/article/reconquerir-la-qualite-de-l-eau-en-regions-de-grandes-cultures-agriculture-biologique-et-reconnexion-avec-l-elevage; **Benoit, M., Garnier, J., Billen, G., Tournebize, J., Gréhan, E. & Mary, B.** 2015. Nitrous oxide emissions and nitrate leaching in an organic and a conventional cropping system (Seine basin, France). *Agriculture, Ecosystems & Environment*, 213: 131–141. https://doi.org/10.1016/j.agee.2015.07.030; Lassaletta *et al.* forthcoming; Quemada *et al.*, unpublished data.

for roughly 1 percent of global GHG emissions (Menegat, Ledo and Tirado, 2022). Systemic changes in dietary patterns, incorporating an increased plant-based diet, together with systemic structural changes, could have a significant impact on future pathways for N use in cropping systems (Billen *et al.*, 2024). Combined with population growth, projected trends toward greater animal product demand across many low- and middle-income countries (LMICs) suggest continued increasing demand for land for feed production and N input requirements across the globe (Gerten *et al.*, 2020; Springmann *et al.*, 2018). Shifts towards more plant-based diets in high-income regions represent an important demand-side complementary action and can reduce the growth of the demand for global animal feed (Billen *et al.*, 2021; Bodirsky *et al.*, 2022; Garnier *et al.*, 2023).

## 2.5 CASE STUDIES

### 2.5.1 Use of nitrification and urease inhibitors in Spain

Ammonia volatilization and N leaching are two major N loss pathways from both environmental (i.e. impacts on ecosystems, indirect $N_2O$ emissions and human health) and quantitative viewpoints (i.e. under favourable conditions both N loss pathways can become notable "holes in the pipe" that connects N inputs and N outputs). These losses occur once organic or synthetic fertilizers are applied to arable crops and grasslands. The use of urease inhibitors such as N-(n-butyl)thiophosphoric triamide (NBPT) with ureic-N-based fertilizers (e.g. urea) has been described as an effective practice to facilitate the incorporation of urea into the soil profile and to decrease $NH_3$ volatilization. Moreover, the temporary decrease of $NH_4^+$ concentrations in the soil can result in lower nitrification rates and mitigation of derived N-loss pathways (i.e. $NO_x$ emission or $NO_3^-$ leaching). Among the enhanced-efficiency fertilizers, the use of nitrification inhibitors is the most effective tool for abating leaching losses and nitrification/denitrification-induced $NO_x$ emissions. Nitrification inhibitors such as dicyandiamide, nitrapyrin and pyrazole-based compounds (e.g. 3,4-dimethylpyrazole phosphate) can be used with ureic-N- and $NH_4^+$-N-based fertilizers (including organic sources such as animal manures) to delay $NH_4^+$ oxidation to $NO_3^-$ through nitrification (by inhibiting the ammonia monooxygenase enzyme involved in the first step of nitrification). As a result, the use of nitrification inhibitors extends the availability of $NH_4^+$-N for plant uptake but could result in pollution swapping through the increased opportunities for $NH_3$ volatilization under high pH conditions (Qiao *et al.*, 2015). The use of a double (or dual) nitrification plus urease inhibitor appears a promising strategy for effectively reducing both volatilization and nitrification rates and the derived reactive-N losses, thereby increasing NUE in croplands. Its use with one of the most widespread synthetic fertilizers worldwide (urea) can mitigate the negative impacts of this fertilizer (reactive N losses, particularly those of $NH_3$) while potentially keeping its advantages of lower upstream emissions than other inorganic N sources, price, availability and storage. In this context, the use of urea with and without a double inhibitor (containing NBPT as urease inhibitor, and DMPSA (2-(3,4-dimethyl-1H-pyrazol-1-yl)-succinic acid) as a pyrazole-based nitrification inhibitor) was evaluated in a three-replicated field experiment in central Spain (semi-arid Mediterranean conditions), in a rainfed winter wheat (*Triticum aestium* "Marcopolo") crop. Gaseous losses ($NH_3$ and $N_2O$), soil mineral N and agronomic response (yield and NUE) were monitored during the 2019/20 cropping season (Guardia *et al.*, 2021).

Results showed that the use of a double inhibitor with urea significantly reduced average $NO_3^-$ concentrations in the soil, leading to significant mitigation of $NH_3$ volatilization (by 51 percent) and $N_2O$ emissions (by 92 percent) compared with conventional urea without inhibitors. Synthetic N fertilization elicited a greater response in N yield than in biomass yield, leading to low agronomic and physiological efficiencies but acceptable crop recovery efficiency. This could be explained by the effect of genotype (wheat variety) by environmental conditions, which cause wheat grown in a Mediterranean climate to have lower yields but higher protein content in comparison with the average global values for wheat (Savin, Sadras and Slafer, 2019). As a result, the use of the double inhibitor resulted in a significant enhancement of crop recovery efficiency (by 36 percent), partial nutrient balance (by 30 percent), and a significant abatement of N surplus (by 38 percent), compared with urea alone (Guardia *et al.*, 2021).

These results are consistent with global meta-analyses. For instance, Abalos *et al.* (2014) obtained a significant increase in NUE with the use of nitrification and urease inhibitors (15 percent), while Sha *et al.* (2020) found a global increase of 50 percent in fertilizer-N recovery through crop uptake. Both reports highlight the quite consistent response and the broad applicability (arable/fruit trees, rainfed/irrigated, humid/dry climates), but some significant effects of soil (pH, texture, organic matter), environmental and management factors (including application rates, split applications and timing) on the agronomic effectiveness. Some unusual examples of inefficacy have been found: (i) when conditions are favourable for low N losses such as leaching, denitrification or volatilization, thus masking the N-loss mitigation effect of the enhanced-efficiency fertilizer; (ii) in field trials with enough or excessive N supply, so the potential for improving N acquisition is limited (Rose *et al.*, 2018); and (iii) under conditions with severe biotic or abiotic stresses which seriously limit crop development.

The increased cost of fertilizer appears to be the main barrier to the widespread adoption of enhanced-efficiency fertilizers. Potential solutions for the economic barriers include economic policy incentives and better awareness of the opportunities for improving crop yield and quality or maintaining the agronomic performance by using lower N-fertilizer rates (through improved NUE), which offsets the extra cost of the fertilizer.

### 2.5.2 Technology-oriented solutions to increase nitrogen use efficiency in intensive cereal grain agriculture in Mexico

The cereal production systems of Northwestern Mexico are characterized by the availability of irrigation water, mainly from reservoirs. Irrigated systems are associated with high use of inputs due to their high yield potential. The Yaqui Valley of Mexico in the state of Sonora is no exception. While there are several crops in the cropping system, such as safflower, maize and chickpeas, it is dominated by durum wheat (*Triticum turgidum*). The NUE of wheat in the Yaqui Valley has been estimated at around 31 percent, the N that is not recovered by the crop has significant environmental consequences. At least three pathways of N loss have been documented in the valley. The first is atmospheric emissions. Matson *et al.* (1998) demonstrated substantial emissions of the potent greenhouse gas $N_2O$ in the Yaqui Valley in an already difficult scenario of climate change and high fertilizer prices. More recently, an $N_2O$ emission factor of nearly 0.5 percent at 260 kg N/ha application has been reported in the region (Millar *et al.*, 2018). The second pathway is runoff from wheat fields, which feeds large phytoplankton blooms (54–577 $km^2$) in the Gulf of California (Beman, Arrigo and Matson, 2005). These algal blooms occur within days of high-rate N fertilization and irrigation in the intensive wheat production fields. The inefficient use of N leads to eutrophication in the Gulf of California. The third N loss pathway is through $NO_3^-$ leaching in wheat fields, as demonstrated by Riley *et al.* (2001). Low NUE results in N losses greater than 60 percent, of which 20–40 percent is lost to surface waters (Riley, Ortiz-Monasterio and Matson, 2001).

The Yaqui Valley is the birthplace of the Wheat Green Revolution, which promoted the use of modern high-yielding semidwarf wheat varieties, irrigation and high input use. This region is responsible for supplying durum wheat for domestic consumption and for export to ten countries (SIAP, 2023). Wheat production in Sonora represents almost 60 percent of the national production and contributes to the country's eleventh place in the world for bread wheat production and its being the third largest exporter of durum wheat (SIAP, 2023).

The following combination of strategies has been used and documented to reduce N losses in the Yaqui Valley.

1. The link between plant breeding and N fertilization has been studied in the region. It is well documented that modern semidwarf wheat cultivars respond more to N fertilization, which translates into higher economic rates and higher returns when N fertilizer is available compared with older cultivars (Ortiz-Monasterio *et al.*, 1997).
2. Increased efficiency of N fertilizer use can be achieved with management practices that account for spatial variability in soil properties and temporal variability in climate. Lobell *et al.* (2004) developed an N management decision model for wheat in the Yaqui Valley that incorporates hypothetical diagnostics of soil N and growing season climate. The model is then used to quantify the potential value of these forecasts concerning wheat yields, farmer profits, and excess N application.
3. The GreenSeeker technology uses an optical sensor combined with a reference N-rich strip and a crop-specific algorithm to make mid-season fertilizer N recommendations. The technology has three basic components: (1) establishing a reference or rich strip where a non-limiting dose of N is applied; (2) using the GreenSeeker sensor to measure the response in the reference strip and the rest of the plot being diagnosed; and (3) plugging the resulting sensor readings into an equation that determines the optimal dose of N to apply to the plot, thus avoiding over-application of N fertilizer. As a handheld tool available at low cost, it can be an appealing option for both extension providers and low- and middle-income farmers for whom more expensive technologies are inaccessible or unaffordable (Lapidus *et al.*, 2022; Ortiz-Monasterio and Raun, 2007).

The following results have been measured through these strategies.

1. From 1950 to 1985, wheat breeding at the International Maize and Wheat Improvement Center (CIMMYT) improved NUE by almost 2 percent per year at high fertility levels in the Yaqui Valley (Ortiz-Monasterio *et al.*, 1997).
2. Nitrogen diagnostic tools and weather forecasts: Soil variability is about three times as important as climate variations as a potential impact on profits in Sonora. The model was used to simulate the effect of increases in fertilizer price, which have similar positive effects on excess N application but negatively influence farm profits. Finally, it was concluded that even limited information on soil or climate can be useful for farm management decisions (Lobell, Ortiz-Monasterio and Asner, 2004).
3. The GreenSeeker: In the Yaqui and Mayo valleys, where the technology has been adopted for the longest time, the GreenSeeker use has led to an average of USD 38 per hectare in additional farmer profits, totalling USD 1.9 million of additional earnings over ten years. It has reduced GHG emissions by an estimated 9.5 tonnes $CO_2$eq (Lapidus *et al.*, 2022).

Strategies to optimize N use have undoubtedly helped reduce application rates, N losses, pollution and emissions. They have allowed producers to spend less on fertilizer. Part of the effectiveness of these strategies is that they have been adapted to the type of soil, climate, crops and varieties and the problems prevalent in the Yaqui Valley. Much remains to be done to improve the implementation, adoption and scaling up of N fertilizer optimization practices. Although technical assistance and the promotion of public and private investment are elements that would contribute to improving the adoption and scaling up of practices, it is the change in the mentality of producers and better governance that would boost the long-term adoption of strategies to make N fertilizer use more efficient. Dissemination of best practices, capacity building, and raising awareness of the linkages between soil management, pollution and climate change can help address the root causes of inappropriate N fertilizer use and the associated impacts.

## 2.6 CONCLUSIONS AND KEY MESSAGES

Improving the management of N across global croplands is essential for sustainable food production, climate stabilization, biodiversity, and improving air and water quality. Croplands play an important role in the flow of N through the agrifood system, as croplands are the entry point for most of the external N inputs into the agrifood system.

Nitrogen use efficiency is an essential indicator for assessing the agro-environmental performance of cropping systems and can be defined as the proportion of N inputs that are retained in agricultural outputs. High-performing cropping systems achieve the combination of high productivity, adequate rotations, high NUE, and low N surplus. Looking across the globe reveals massive variability between countries in their NUE and yield performance. Many factors influence country NUE, including the local and regional crop mix, agronomic technologies, and management practices. As agricultural sectors evolve, each of these factors determines national NUE.

Many strategies now exist for improving NUE from field to farm to society. These include a set of recommendations for applying N fertilizers with the right rate, right placement, right time and right source (4Rs) that can be fostered by precision farming and remote sensing and reinforced by other management practices and technological innovations aiming to improve crop status. Diversifying crop rotations and increasing leguminous crops represent additional integrative approaches to increasing NUE while reducing regional pollution. Furthermore, implementing agroecological practices such as strip cropping, cover cropping and conservation agriculture can increase soil nutrient status and health and minimize N losses. Moreover, sustainable N management should not be focused solely on ensuring N crop status and fertilization but on ensuring soil health and include (regenerative) practices to ensure healthy soils.

The fate of N in manure as a pollutant or as a useful source of N fertilizer is a major determinant of agrifood sustainability, and a variety of efforts related to spatial planning, redistribution of livestock, and reduction of livestock numbers in densely animal-populated areas, to improve crop and livestock integration, can improve NUE. Other systemic approaches include examining the optimal allocation of N resources across scales, the use of human excretion, and demand-side approaches that address consumption patterns. Regardless of the specific strategies adopted, historical and contemporary cropland N budgets provide a foundation from which to assess more sustainable N futures.

# Chapter 3
# Nitrogen use efficiency in livestock systems

## 3.1 INTRODUCTION

The livestock sector plays an essential role in the livelihoods of rural people and contributes to food security and nutrition. With the growing demand for terrestrial animal source food (TASF), livestock production has been growing rapidly (Gerber *et al.*, 2013). In middle- and high-income countries, small-scale mixed systems have transitioned towards medium and large operations that are characterized by increased efficiency, productivity, and high outputs (Gerber, Vellinga and Steinfeld, 2010; Tullo, Finzi and Guarino, 2019). This intensification of livestock systems in middle- and high-income countries has been accompanied by the use of concentrate feeds (cereal grains, soybeans) and shifting away from open-range feeding and backyard systems (Bouwman *et al.*, 2013; Ramankutty *et al.*, 2018). Feed production has expanded in countries with a large availability of land, mainly in Latin America and Northern America, and is exported to countries with geographical concentrations of livestock farms. The economies of scale, globalization, and internationalization of livestock supply chains have been fuelled by the availability of low-cost labour, investment in agricultural assets, favourable domestic policies, development of cold chains, and the efficiency of maritime transport (Gerber, Vellinga and Steinfeld, 2010).

This expansion of livestock systems has resulted in a disruption of N cycles on a global scale. In regions where feed crops are grown, the demand for synthetic N fertilizer has increased and resulted in their excessive application, leading to high N losses in the environment. Regions with a high concentration of livestock rely on the import of N-rich feed, which, when ingested by animals, is partially converted into animal products. The remaining N is excreted in manure. This manure is often poorly managed at the farm level (Bai *et al.*, 2016; Gerber and Menzi, 2006). In many cases, it cannot be recycled effectively due to the unavailability of agricultural land in livestock-dense areas or the high cost of transport. Thus, in some countries, manure is discharged into water bodies or overapplied to limited agricultural lands. Despite these changes, livestock systems in low-income countries are still characterized by small-scale producers and pastoralists, who depend on low input levels and biological N fixation to access N from grazing natural grasslands and use crop residues and food leftovers as feed.

Feed demand for the global livestock sector was estimated to be 6 billion tonnes in 2010 (Mottet *et al.*, 2017). About 40 percent of global available arable land is used to grow feed materials, and 15 percent of terrestrial land is used for grazing (Mottet *et al.*, 2017). The global production of TASF is one of the main drivers of N losses in the total agricultural system (Galloway *et al.*, 2010; Sakadevan and Nguyen, 2017). Livestock consumed about 106.9 Tg N/yr in animal feed in 2010, of which almost 90 percent was excreted in urine and faeces (Uwizeye *et al.*, 2020). The excreted N is vulnerable to losses via $NH_3$ volatilization, denitrification, and leaching during collection, storage, treatment, and following application to cropland.

This chapter focuses on NUE in livestock agrifood systems, explaining how NUE can be quantified in livestock systems (section 3.2) and outlining the major factors influencing NUE (section 3.3). Section 3.4 provides an overview of N use and challenges in major livestock systems, and section 3.5 highlights a global perspective on N use and flows. Section 3.6 presents different case studies on NUE in livestock systems and lastly, a summary of the chapter with key points is given in section 3.7.

## 3.2 QUANTIFICATION OF NITROGEN USE EFFICIENCY IN LIVESTOCK SYSTEMS

Nitrogen use efficiency provides insight into the efficiency of N use at animal, herd, farm or regional level, depending on the chosen system boundary. The NUE indicator can indirectly indicate potential environmental pressures but does not account for the environmental impacts associated with N losses. In the context of global livestock systems, NUE has been expanded to the supply chain level, referred to as life-cycle-NUE (Suh and Yee, 2011; Uwizeye *et al.*, 2016). This indicator considers N inputs, reuse, recycling, changes in stocks, and the efficiency of new N inputs recovery in final products.

A review by Gerber *et al.* (2014) revealed that NUE at animal level varies across different livestock categories: 15–35 percent for dairy cattle, 4–8 percent for beef cattle, 10–44 percent for pigs, and 25–62 percent for poultry. At the farm or system level, considering all livestock species, NUE ranges between 5 and 45 percent. These indicators are further discussed in the subsequent sections. Recent estimates of NUE in the livestock sector in Europe found high values of NUE at the farm-system level. For instance, Hutchings *et al.* (2020) found that NUE for dairy production systems was in the range 54–55 percent and 44–62 percent for ruminant meat systems on unconstrained land. For dairy systems, Uwizeye *et al.* (2020) found NUE at the animal production stage, ranging from 69 to 84 percent. These differences are due to the consideration of manure in the outputs due to its fertilizer value.

FAO, through the Livestock Environmental Assessment and Performance (LEAP) Partnership, has developed a comprehensive guideline to quantify nutrient flows and associated environmental impacts in livestock systems (FAO, 2018a). The guidelines provide a step-by-step approach to analyse N and P flows in livestock and crop-feed systems, impact assessment methods, and guidance to interpret the results. The guidelines have been applied in different case studies in different regions and scales (Hutchings *et al.*, 2020; Löw, Karatay and Osterburg, 2020; Uwizeye *et al.*, 2020).

## 3.3 FACTORS INFLUENCING NITROGEN USE EFFICIENCY IN THE LIVESTOCK SECTOR

### 3.3.1 Animal genetics and breeding

As N is an essential building block for animals, part of the N taken up through the diet is retained to form proteins and nucleic acids. Proteins are necessary for the maintenance, growth and production of animal products. Animal genetics plays an important role in increasing NUE. Through breeding and breeding improvement technologies, animals with a high feed conversion efficiency can be selected. Animal productivity can be increased substantially through improved feed quality, feeding management, and improved animal health, which can decrease the amount of N excreted through manure (Sutton *et al.*, 2011).

### 3.3.2 Animal physiology and categories

The efficiency with which N in feed is converted into animal products is an indicator to determine how much of the N intake is used for production purposes. The feed conversion ratio (FCR) of livestock categories is used as an indicator to measure how much feed is needed to produce a unit of product. This is largely dependent on the animal's physiology, health, performance, and feed quality. Feed conversion ratios differ greatly between animal species and categories (Mekonnen *et al.*, 2019). Peters *et al.* (2014) calculated FCRs of different livestock categories on different per-unit bases (kg feed dry matter [DM] per kg output) and protein content (kg feed DM/100 g). The latter is particularly needed to determine the NUE of a given livestock category. Although the study showed substantial variation between livestock categories and per-unit bases, overall cattle for beef production showed the lower FCR and excretion of a large proportion of N, resulting in low NUE. Chicken and pigs show a higher FCR, with layer hens showing the highest efficiency in terms of protein content (kg feed DM/100 g) compared with broilers and pigs. Low NUE is typically associated with high losses of N through $NH_3$ volatilization, with an average of 20 percent of ingested feed N lost as $NH_3$ across livestock categories (Groenestein *et al.*, 2019).

It is important to not only look at animal physiology and the feed conversion efficiency to determine its influence on NUE. Land use for feed production and accounting for the proportion of human-edible and inedible feed in the diet alters the efficiency of livestock categories (Peters *et al.*, 2014; Wilkinson, 2011). Cattle for beef and milk typically have a low NUE, but when correcting for inedible human feed, its feed conversion efficiency increases substantially. On the other hand, when looking at the edible protein FCR, Wilkinson (2011) found that all monogastric species have a value > 1, meaning that more human-edible protein is consumed by the animal than protein produced in the end product (Table 1).

TABLE 1
**Feed conversion ratios (total protein in feed and human-edible protein in feed) for different livestock categories**

| | Total feed protein (kg/kg edible protein in animal product) | Human-edible protein (kg/kg edible protein in animal product) |
|---|---|---|
| **Milk** | 5.6 | 0.71 |
| **Upland suckler beef** | 26.3 | 0.92 |
| **Lowland suckler beef** | 23.8 | 2.0 |
| **18–20 months beef** | 14.9 | 1.6 |
| **'Cereal' beef** | 8.3 | 3.0 |
| **Average (upland/lowland) lamb** | 33.0 | 1.4 |
| **Pig meat** | 4.3 | 2.6 |
| **Poultry meat** | 3.0 | 2.1 |
| **Eggs** | 3.2 | 2.3 |

*Note:* Suckler beef refers to calves reared with their mother. Cereal beef are cattle fed on a diet with a high proportion of cereal crops.

Total feed protein indicates how much dietary protein in feed is used to produce human-edible animal protein in milk, meat and eggs. Human-edible protein indicates how much dietary protein in feed suitable for direct human consumption is used to produce human-edible animal protein in milk, meat and eggs. A value > 1 indicates that more protein was consumed than produced.

*Source:* Adapted from **Wilkinson, J.** 2011. Re-defining efficiency of feed use by livestock. *Animal*, 5(7): 1014–1022. https://doi.org/10.1017/S175173111100005X

### 3.3.3 Livestock feed production

The significant contribution of livestock to environmental impacts is largely due to feed production to sustain the global demand for TASF (Bouwman *et al.*, 2013; Galloway *et al.*, 2010). Feed production encompasses the production of fodder, grains, cereals, and other concentrates, as well as by-products from crop production. On a global level, the main feed materials used for livestock production are grasses and leaves (46 percent) and crop residues (19 percent). Feed materials suitable for human consumption account for about 14 percent of livestock feed (Mottet *et al.*, 2017). It is estimated that about 80 percent of the N harvested in crops is used to feed livestock (Sutton *et al.*, 2013). Intensive systems are characterized by diets rich in soybeans, grains, and other supplements to maximize production efficiency.

In many parts of the world, feed crops (including grains and cereals) are grown with high N inputs that exceed the N requirements of the plant. The current mean global efficiency with which N is recovered in (food and feed) crops is 48 percent (Quan *et al.*, 2021; You *et al.*, 2023), whereas for cereals, the global average is 35 percent (Omara *et al.*, 2019). It is estimated that around 70 percent of N that enters the crop system from fertilizer is lost to the environment (Galloway *et al.*, 2010) via leaching and runoff in soils and air through the volatilization of $NH_3$ and other N gases. Hence, feed crops are associated with these losses. When feed is exported, N ends up in high-density livestock regions, where a large part of this N is excreted in manure. In intensive systems, largely dependent on external (concentrated) feed inputs and with limited access to cropland, N excretion in manure is accumulated. If not properly managed, substantial manure N losses to the environment occur. Feed production and associated intensive livestock systems depending on external input of concentrated feed, create a disrupted N cycle, with excessive losses of N to the environment from high synthetic N fertilizer input and accumulation of manure N.

### 3.3.4 Manure management

How manure is managed and processed greatly influences N losses, and proper manure management plays an important role in the overall NUE of livestock systems. Most of the N in feed ingested by animals is excreted through manure (faeces and urine). In many parts of the world, poor manure management systems result in the loss of N compounds (Uwizeye *et al.*, 2020). Nitrogen emissions from manure occur in housing systems, storage, and during the application to grasslands and croplands, as well as during grazing (Oenema *et al.*, 2008; Rivera and Chará, 2021). Manure contains organic-bound and mineral-bound N. Organic-bound N can be lost through $NH_3$ and $N_2O$ emissions during the breakdown processes of organic N by urease enzymes (Sigurdarson, Svane and Karring, 2018). The extent to which $NH_3$ emissions occur depends on the type of manure, temperature, moisture content and pH. In housing systems where urine and faeces are combined and stored as slurry, significant $NH_3$ emissions occur when the urea in urine is converted to $NH_3$ by the urease enzyme in the faeces fraction. Nitrous oxide, a potent GHG, is formed from organic-bound N. There are still uncertainties about the magnitude of these emissions from manure (Amon *et al.*, 2006). Nitrogen is lost through leaching of $NO_3^-$ during storage on sites in direct contact with soils or following unregulated disposal of manure into the environment. Measures to increase sustainable manure management are further discussed in section 3.4.

### 3.3.5 Rangelands and grasslands management

Grasslands play a significant role in ruminant production systems around the world. A large part of grasslands is overused and poorly managed due to the high demand for forage production for both grass-based milk (27 percent) and beef (23 percent) production (Conant, Paustian and Elliott, 2001). Grasslands have a high inherent SOM content, which supplies plant nutrients, limits soil erosion, and increases their water-holding capacity. These positive aspects are counteracted if they are poorly managed. For instance, overgrazing in ruminant systems with a high stocking rate, can result in soil degradation and N stock decline. As a result, these grasslands degrade and lose their productivity and capacity to sustain livestock herds. On the other hand, grassland in intensive ruminant grazing systems can receive high amounts of nutrients from deposited faeces and urine, exceeding the capacity of grassland soils to hold these nutrients for grass production. This results in leaching of $NO_3^-$ and emissions of $NH_3$ and $N_2O$. Sound rangeland and grassland management can, therefore, contribute to balancing N cycling and minimizing N mining or N losses (see section 3.4.1).

## 3.4 REVIEW OF NITROGEN USE AND CHALLENGES IN DIFFERENT LIVESTOCK SYSTEMS

Nitrogen use in livestock systems heavily depends on the type of livestock species and the system in which the animals are kept. Solutions to increase NUE and minimize losses to the environment differ across various livestock systems and depend on the specific challenges farmers encounter. This section considers the major livestock systems and describes N challenges and potential solutions for selected production systems.

### 3.4.1 Grassland-based systems

Ruminant species (e.g. cattle, sheep, goats) kept in grassland-based systems are characterized by a wide variety of farm characteristics, which can be subdivided into pastoral, extensive and intensive grazing systems. The following sections describe the general characteristics of these subsystems and their challenges and possible solutions to increase NUE.

***Pastoral systems***

Pastoral systems are a form of mobile grazing system predominantly found in drylands of Africa, Asia, Australia and some parts of Eastern Europe (FAO, 2021a). Pastoralism relies on marginal lands often unsuitable for crop production. Marginal lands are defined as less favourable agricultural areas characterized, among other factors, by limited agriculture potential, resource degradation, and low productivity (Ahmadzai *et al.*, 2022). These systems are characterized by the daily and seasonal movement of livestock, where animals grazing during the day are moved to drink water in the morning and evening and are sometimes housed in enclosures overnight (Carbonell *et al.*, 2021). This results in the translocation of N as animals move to pastures, enclosures and waterholes. In enclosed areas, manure accumulation is high and can lead to N accumulation and loss. In Kenya, for example, manure accumulated in enclosures (*bomas*) is collected by crop farmers and added to their croplands (Carbonell *et al.*, 2021). During the dry season, pastoralists rely on crop residues to feed their livestock and thus cooperate with crop farmers who benefit from the manure excreted on their croplands. Pastoral systems working together with crop systems and the exchange of resources during the dry season result in a balanced system where grasslands in dryland regions are managed sustainably. Challenges within pastoralist systems include the intensification of crop production that is increasingly dependent on synthetic fertilizers (Kasymov *et al.*, 2023). Manure decreases in value for crop farmers, and grazing of crop residues during the dry season becomes either unavailable or only possible with payment. As a result, overgrazing of grassland regions occurs, which depletes soils, decreases nutrient cycling within the system, and eventually degrades grasslands and rangelands. Globally, rangelands affected by degradation are estimated to be about 50 percent (UNCCD, 2024). In low- and middle-income regions, where drought periods are prolonged due to climate change, degraded grasslands and rangelands are not able to sustain productivity. Furthermore, if cattle are kept on pastures for prolonged periods, cattle trampling can cause soil compaction and reduction of soil fertility and productivity of grasslands (Hamza and Anderson, 2005). It is essential that these systems are managed properly to sustain soil health and productivity and mitigate detrimental effects on the environment.

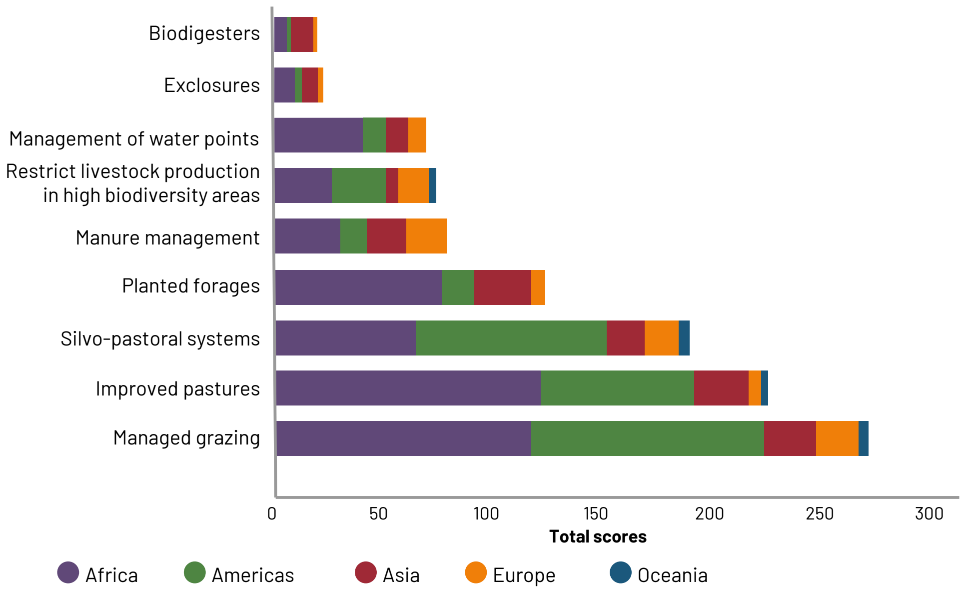
As the use of synthetic fertilizers is sometimes subsidized and stimulated, policy incentives could play an important role in restoring the pastoral and crop production systems. Sustainably managed grasslands not only increase nutrient cycling in the agricultural system but increase resilience to climatic changes through healthy soils and fodder production. A study by Carbonell *et al.* (2021) showed that major N losses in pastoral systems occur through the leaching of $NO_3^-$ and $NH_3$ emissions from manure accumulation in enclosures. Different management strategies enhance N cycling, minimize N hotspots and related leaching to the environment, and increase soil health and productivity. A key strategy is to prevent overgrazing and nutrient mining by identifying how much livestock can be supported by a specific grazing area, known as the stocking rate (Carbonell *et al.*, 2021; Kasymov *et al.*, 2023). The optimum stocking rate can be determined by estimating the number of livestock that can be sustained by the available natural resources necessary to meet their energy requirement needs. With this, an estimation can be made of the amount of N manure that will be returned to the soil to enhance fertility. Rotational and seasonal grazing and resting of pastures allow vegetation to recover and plant growth and productivity to increase. Further prevention of nutrient losses can be realized through improved manure management. Pastoralists who enclose their animals at night can increase the mobility of the enclosures and limit the period animals stay in the same enclosure to decrease N losses from there (Carbonell *et al.*, 2021; Fenetahun and Xu Xinwen, 2018).

***Extensive and semi-extensive systems***

There is a vast variety of extensive and semi-extensive systems, including cattle kept for beef production in Latin America (Jaurena *et al.*, 2021; Nin, Freiría and Muñoz, 2019); small ruminants and cattle production systems where animals are grazed on marginal lands, including grasslands and rangelands (Oosting *et al.*, 2022); and smallholder systems that either focus solely on livestock production or are combined with crop production (Oosting *et al.*, 2022; Paul *et al.*, 2021).

When grazing is the primary feeding strategy, overgrazing can become a challenge. Overgrazing leads to plant degradation from the mining of soil nutrients, which ultimately results in the depletion of soil N stocks (McSherry and Ritchie, 2013). Improving N cycling in pastures should focus on improved grazing management, such as rotational grazing, which can be done in various ways with different stocking rates and seasonal or rotational frequencies (Franke and Kotzé, 2022; Rueda *et al.*, 2020; Teutscherová *et al.*, 2021). Improved manure application and increased use of N fertilizers can contribute to improving soil health, replenishing depleted soils, and improving nutrient cycling (Alecrim *et al.*, 2023; Cardenas *et al.*, 2019; Dos Santos Cordeiro *et al.*, 2021). Improving pastures and grazing management play a key role in enhancing the sustainable development of livestock in grassland-based systems (Paul *et al.*, 2021), as shown in Figure 12. Moderate grazing intensities have been shown to increase the NUE of the livestock system by 6.2 percent as compared to intensive grazing (Xu, Jagadamma and Rowntree, 2018), and improved

FIGURE 12
**Results from a livestock expert survey on promising livestock practices and focus areas**




*Note:* Survey was conducted by a group of 260 livestock experts, selecting the most promising sustainable livestock management practices and focus areas. Respondents scored the three most important options provided. Total scores were derived by taking the sum of responses selected as "most important" (three points), "second most important" (two points), and "third most important" (one point).

*Source:* Authors' elaboration based on **Paul, B.K., Butterbach-Bahl, K., Notenbaert, A., Nduah Nderi, A. & Ericksen, P.** 2021. Sustainable livestock development in low- and middle-income countries: shedding light on evidence-based solutions. *Environmental Research Letters*, 16(1): 011001. https://doi.org/10.1088/1748-9326/abc278

manure management and application in smallholder systems can increase overall NUE by 5–20 percent (Rufino *et al.*, 2007). See section 3.6.3 for a case study assessing N flows and NUE of dairy systems in East Africa.

#### *Intensive grassland-based systems*

Livestock systems in intensive grassland systems have a higher stocking density, and grasslands are more intensively managed than livestock systems in extensively managed systems. Practices may include rotational grazing and external N inputs through fertilization. Countries where these systems are among the most prevalent are New Zealand and Ireland (Luo and Ledgard, 2021; O'Donovan, Hennessy and Creighton, 2021). Grassland-based systems are characterized by low inputs of external feed, with pasture intake representing 80–90 percent of total feed intake. In the last decades, grazing systems have intensified with an increased demand for external inputs such as synthetic N fertilizers and feed. These practices are linked to an increase in N losses and an overall decrease in NUE (O'Donovan, Hennessy and Creighton, 2021). Livestock kept in grassland-based systems can have a high protein intake as the grass is consumed in an early physiological growth stage, which is high in degradable protein. This can result in an excess of dietary N intake, which will be excreted mostly in urine. To increase NUE from grassland-based systems, the focus should be on balanced N fertilization, grazing or harvesting the grassland at a later physiological growth stage, and the use of low-protein supplementary feed (Sutton *et al.*, 2022). Maintaining grasslands rich in leguminous mixtures for increased biological N fixation reduces the need for external N inputs. If unfertilized, grasslands have the potential to prevent the leaching of $NO_3^-$ to neighbouring water bodies and can be used as buffers to prevent losses of N to natural land or streams (Sutton *et al.*, 2022). Grasslands, both grass–clover mixtures and predominantly grass pastures, can play an important role in reducing losses of N due to their increased capacity of N storage in plant biomass and litter. Perennial grass pastures have a longer N uptake period than annual crops. Overall, permanent grasslands increase soil N (and C) and increase N retention capacities (Sutton *et al.*, 2022).

### 3.4.2 Smallholder zero-grazing systems

A prevalent livestock system in LMICs is smallholder zero-grazing, where cattle and small ruminants are kept in enclosed areas and have limited or no access to pastures. Animals are fed with fodder crops and concentrates either grown on agricultural land or harvested from communal areas through cut-and-carry. Through continuous harvest of fodder and insufficient returns of N to soils, these systems are linked to low soil fertility and nutrient depletion.

Furthermore, manure is poorly managed or either uncollected, collected and stored as uncovered heaps or discharged into the environment (IAEA and FAO, 2008; Ibrahim, Graham and Leitner, 2021; Rufino *et al.*, 2007; Teenstra *et al.*, 2014). As a result, significant amounts of N are lost through emissions and leaching. Improving NUE and minimizing losses of N to the environment should focus on improving manure management systems, promoting manure composting and recycling to cropland, and high-quality fodder production.

Minimizing storage time significantly reduces N losses from manure and, if direct land application is possible, this should be prioritized as the first management practice. Direct manure application is only possible when vegetation is present to take up the N from manure. In smallholder farms, different storage techniques have been found (Teenstra *et al.*, 2014). If stored in a heap, without cover or hardened floor, N losses will occur through emissions and leaching. Storage units with a concrete floor can significantly reduce $NO_3^-$ leaching if the leachate is collected. Additionally, roofing or covering the manure heap with a plastic sheet has been found to substantially decrease the amount of N that is lost from manure and increase the overall N cycle of smallholder farms (Rufino *et al.*, 2007; Tittonell *et al.*, 2010). The application of manure stored in covered pits has been shown to increase crop yields significantly compared to uncovered, stored manure (Mutiro and Murwira, 2004). The adoption of improved manure management and storage techniques should be reinforced by supporting manure policies, which are often lacking in LMICs (Ndambi *et al.*, 2019).

A manure treatment technique that could be adopted by smallholder farms is the anaerobic digestion of manure (Ndambi *et al.*, 2019) to produce biogas as a renewable source of energy for cooking or lighting in rural households. The digestate that remains after the digestion process can be used as organic fertilizer, which could reduce synthetic N fertilizer input. The installation of a digester requires a high initial investment and, for many smallholder farmers, requires technical and financial support through subsidies (Ndambi *et al.*, 2019).

### 3.4.3 Backyard monogastric systems

Monogastric animals kept in backyard systems form an important source of livelihood in many regions of the world. These systems are characterized by family farms keeping a small number of animals that provide an important source of food. In the tropics, the majority of TASF is produced through these small-scale farms, contributing to food security and nutrition for the rural population (Herrero *et al.*, 2010). Pigs and poultry held in backyard systems typically scavenge their food and are supplemented with household leftovers (Oosting *et al.*, 2022). These systems are characterized by low external inputs, and on-farm labour is relatively low. Manure is typically collected and spread on cropland or is sometimes disposed of in the environment.

Nitrogen use efficiency of backyard systems is relatively high when compared to other livestock systems, with averages ranging from 35 to 45 percent for backyard pig and poultry systems (Uwizeye *et al.*, 2020). There is a trend of intensification in these systems, as population growth and higher demand for TASF drive farmers to increase their production. This shift to medium-sized farms is linked to a greater use of external inputs and, subsequently, an increased risk of N losses to the environment. Measures to recycle nutrients in these systems and minimize losses to the environment should focus on sound manure management practices, where manure collection is maximized, and storage facilities minimize losses through leaching and emissions. As a result, manure can be a valuable fertilizer for croplands in smallholder systems, increasing yields and minimizing dependency on external inputs. Other measures to increase NUE on farm level are to reduce FCR (i.e. the amount of feed needed to produce a unit of product), balance protein intake from feed, and improve animal performance (Ma *et al.*, 2021).

### 3.4.4 Intensive dairy and feedlot systems

Intensive cattle systems can be subdivided into sub-systems:

- **Intensive dairy systems** vary across regions but are characterized by mixed farms where feed is composed of forage and is supplemented with maize silage and other concentrated feed, either produced on-farm or purchased from external markets. Dairy cows can be kept in housing systems and partially grazed (either seasonally or year-round). Ammonia emissions occur in housing or yards and manure management systems because manure is often stored as slurry under the slatted floor of the barn. Increasing the grazing days per year of housed dairy cattle is an effective measure to decrease $NH_3$ emissions due to rapid infiltration of urine and plant uptake. This measure could decrease emissions by up to 50 percent if all-day grazing were implemented (Sutton *et al.*, 2022).

  For dairy cows, research has shown that lowering the protein content in the diet can lower $NH_3$ emissions by 46 percent and $N_2O$ by 20 percent compared to the average emissions per dairy cow without decreasing animal health and productivity (Schrade *et al.*, 2023). Overall, NUE in dairy cows could be increased by 25 percent through this feeding strategy (Chowdhury, Wilkinson and Sinclair, 2023). This measure is most applicable to housed animals, as the protein content in their diet is better controlled and more easily managed. Farmers play a key role in adopting these measures as the main stakeholders in animal feeding management.

Tan *et al.* (2023) conducted a comprehensive study on farmers' decision-making processes on animal feeding. Their study found that 65 to 85 percent of farmers did not know of low protein feeding and that feeding strategies were mainly influenced by suppliers. Concerns around low protein feeding mainly revolved around potential negative impacts on animal productivity, animal health, and overall farm performance. Positive impacts such as reduced feeding costs and emissions and improved animal health were recognized. The study underlines the importance of access to information about the benefits of low protein feeding and training programmes.

- **Feedlot systems** are characterized by the fattening and finishing stages of beef production and are dominant in Northern America, Latin America and Oceania (Cowley *et al.*, 2019). In these systems, many beef cattle are kept in open pens with a high stocking density. Feedlots rely on imported feed. The diet consists mainly of silage, cereals (maize, barley and wheat) and soybean meal. Manure is typically accumulated in the pens or stockpiled, which forms the main source of N losses from these systems through $NH_3$ and $N_2O$ emissions (Cowley *et al.*, 2019). Over 80 percent of N fed to the animal is excreted in urine and faeces, of which 67 percent is lost through $NH_3$ emissions (Kissinger *et al.*, 2007), resulting in an NUE of about 20 percent at animal level (Koenig, McGinn and Beauchemin, 2013). In combination with large amounts of imported feed associated with high synthetic fertilizer inputs, feedlot systems have a relatively low NUE. Measurements to reduce N losses from feedlots are primarily linked to reducing protein-N content in diets and improved manure management (Cowley *et al.*, 2019; Galles *et al.*, 2011). Nitrogen losses are largely driven by the spatial disconnection between animals and feed production, and recycling of manure in livestock-dense areas is linked to high transportation costs to croplands (Uwizeye *et al.*, 2020). The on-farm measurements described above will therefore not lead to significant improvements of the life-cycle-NUE of the livestock supply chain.

Feeding a low-protein diet is seen as a cost-effective method to decrease N losses and increase NUE. Bittman *et al.* (2014) and Sutton *et al.* (2022) give a comprehensive overview of feeding measures that can be taken for different livestock species, indicating that for each percent decrease in protein content, $NH_3$ is decreased by 5–15 percent, while reducing $N_2O$ emissions (Sutton *et al.*, 2022), and for increases in overall NUE (Hristov *et al.*, 2015; Koning, Evers and Šebek, 2021; Schrade *et al.*, 2023). Sutton *et al.* (2022) have provided detailed guidelines for feeding measures per livestock species to reduce N losses and increase efficiency. Manure management focused on minimizing emissions and nutrient losses plays an important role in decreasing N emissions from dairy and feedlot systems. Manure is a valuable resource of N (and other nutrients) and can decrease losses of N to the environment as compared to synthetic fertilizers (see section 3.6.2 for a case study on the use of dairy slurry as fertilizer).

### 3.4.5 Industrial pig systems

Pigs held in industrial systems are characterized by large numbers of animals housed in groups, typically on (partly) slatted floors. Pig diets contain high amounts of grains and soybeans and are indirectly linked to N losses from synthetic fertilizer input for feed production. Increasing the use of agrifood by-products and food losses and waste (FLW) could decrease the dependence on imported feed, potentially increasing NUE and reducing N losses (Uwizeye, 2019). According to Parfitt *et al.* (2010), FLW is the "wholesome edible material intended for human consumption, arising at any point of the food supply chain that is instead discarded, lost, degraded or consumed by pests". (See section 3.6.1 for a case study outlining the potential to use FLW as feed.) On average, 60–70 percent of N fed to pigs is excreted through manure. Feeding strategies that focus on lowering the crude protein content in the diet have been proven to have significant effects on reducing N in manure, resulting in lower N losses (Sajeev *et al.*, 2018). Reductions of over 30 percent have been measured from pig housing systems when implementing this dietary measure (Le Dinh *et al.*, 2022).

Manure and urine are collected and stored as slurry, either under the slatted floors or in lagoons. The storage of slurry for long periods is linked to substantial $NH_3$ and $N_2O$ emissions (Kupper *et al.*, 2020). Measures to decrease emissions from manure include frequent removal combined with anaerobic digestion, acidification of the slurry, applying additives and covering lagoons with different types of cover, such as impermeable or permeable synthetic cover, a concrete lid, plastic tiles or natural covers (peat, straw, wood chips, etc.) (Peterson *et al.*, 2020; Sajeev, Winiwarter and Amon, 2018; VanderZaag *et al.*, 2015). Storing urine and faeces separately through housing systems that separate the waste from the moment of excretion or through a solid–liquid separation after excretion can decrease N emissions substantially, too, if low-emission processing and storing techniques are applied for the solid fraction (Sutton *et al.*, 2022). The solid fraction contains most of the P excreted by the animal and has a relatively high organic matter content, whereas the liquid fraction contains most of the mineral N (available from crop uptake) and K. The separated fractions have specific properties and can be applied to agricultural land according to specific soil and crop needs and thus enhance more balanced fertilization.

To minimize losses from storage of manure (fractions), storage in enclosed silos until application is preferred.

Manure (both slurry and liquid fraction after separation) can be used for anaerobic digestion to produce biogas that can be converted to electricity or renewable fuels. The digestate is a by-product that can be used as organic fertilizer. Digestate typically has a higher organic-bound N content and low dry matter content, which increases crop NUE. The organic-bound N can increase $NH_3$ emissions during storage and field application if appropriate measures are not taken (e.g. covered and/or airtight storage and low-emission application) (Sutton *et al.*, 2022). Digestate can be processed further to create fertilizer products that enhance nutrient recovery and crop nutrient utilization (see Figure 13). These techniques include stripping manure, pelleting and concentrating, producing solid and liquid manure fractions, organic fertilizer pellets, and mineral concentrates.

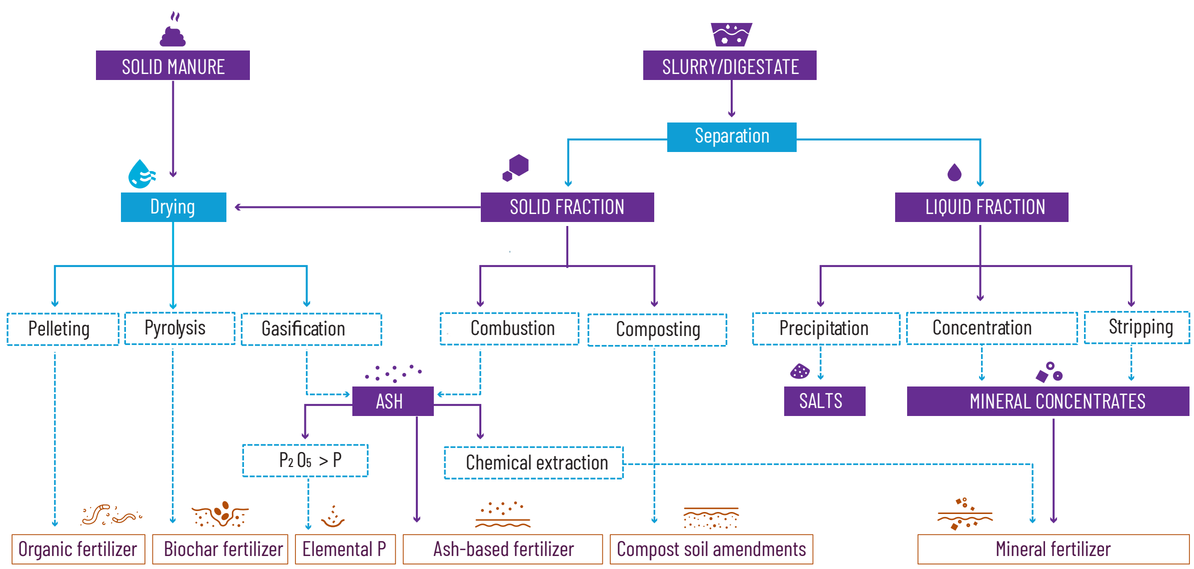
### 3.4.6 Intensive poultry systems

Poultry systems encompass both broiler systems for meat production and layer systems for egg production. Broilers and layers are bred for their efficient production. They are housed in indoor systems and fed mainly a concentrated diet composed of grains (maize, wheat, barley), soybean meal, minerals and vitamins. The intensive poultry sector has increased significantly over the past decades and plays an important role in producing high-quality TASF. It is associated with significant N losses and contributes approximately 15 percent to total N losses from the livestock sector (Uwizeye *et al.*, 2020). Most of the N emitted by poultry is in the form of $NH_3$ (and to a lesser extent $N_2O$) emissions from manure storage. Moreover, poultry systems are associated with indirect $N_2O$ emissions related to the production of concentrated feed.

Poultry manure contains high amounts of uric acid, which can easily form $NH_3$ through hydrolysis. Keeping solid manure stored in dry conditions reduces N emissions and leaching. Ammonia emissions from housing systems of both broiler and layer systems can be minimized with several measures, as summarized by Sutton *et al*. (2022). Frequent removal of manure from the housing system to an enclosed external storage unit reduces N losses in housing systems. Dry storage of poultry manure can reduce leaching of N, while quickly drying solid manure after removal from the housing system prevents $NH_3$ formation from uric acid. Additionally, air scrubbers in housing systems can capture $NH_3$ and improve the air quality of the housing system.

Regarding lowering N losses through feeding, as with ruminants and pigs, poultry show significant reductions when fed diets with low protein content and balanced

FIGURE 13
**Options to combine treatment and processing techniques of manure**




*Note:* Treatment and processing techniques are applicable for pig and cattle manure.

*Source:* Authors' elaboration based on **Sutton, M., Howard, C., Mason, K., Brownlie, W. & Cordovil, C.** 2022. *Nitrogen opportunities for agriculture, food & environment.* UNECE guidance document on integrated sustainable nitrogen management. UK Centre for Ecology & Hydrology. https://unece.org/sites/default/files/2022-11/UNECE_NitroOpps%20red.pdf amino acid requirements (Brink *et al.*, 2022; Malomo *et al.*, 2018; Musigwa *et al.*, 2020; Wiedemann *et al.*, 2016). This solution is crucial for farmers feeding poultry a high-protein diet that exceeds the nutritional requirements of the animal, resulting in decreased NUE and potentially higher feeding costs. Table 2 summarizes the effect of low-protein diets on poultry performance based on the review of Malomo *et al*. (2018). Decreasing the crude protein content of the diet may necessitate supplementation of amino acids (depending on the specific diet formulation and animal needs) to meet animal requirements without compromising animal health and performance (Malomo *et al.*, 2018).

### 3.4.7 Camelid production systems

Camelid production systems can be found on different continents, depending on the camelid species. Bactrian and dromedary camel populations are primarily present in Africa and Asia, whereas llamas and alpacas are found in South America (Zarrin *et al.*, 2020). The largest population of camels is found in East Africa, where animals are mainly kept for milk and meat. Traditionally, camels are kept in migratory pastoralist systems but are found in (semi-)sedentary pastoralist and mixed farming systems as well. In South America, camelid species are primarily kept for fibre production and play an important role in the livelihoods of communities in the Andean highlands. There is little information on N inputs, outputs and losses in camelid production systems and NUE and improvement pathways for these systems have not yet been developed. The resilience of camelid species to live in harsh climatic conditions has been increasingly seen as an adaptive strategy to climate change to ensure food production and security (Rahimi *et al.*, 2022; Wako, Tadesse and Angassa, 2017; Zarrin *et al.*, 2020). Camelid species require less protein in their diet for production and research has shown that camel housing systems emit less $NH_3$ than dairy cattle systems (Nadtochii *et al.*, 2018; Rahimi *et al.*, 2022; Smits and Montety, 2009).

## 3.5 ASSESSMENT OF NITROGEN USE EFFICIENCY AND FLOWS IN LIVESTOCK SUPPLY CHAINS

The livestock supply chains vary from local to global scale. While the production of most TASF in LMICs is still localized, livestock supply chains have increasingly become lengthier and more internationalized. Decoupling of the production stage has caused several challenges to sustainable N management. Understanding cascading impacts on N across the entire value chain is crucial to ensure informed policy and decision-making to reduce pollution while promoting measures to enhance NUE. The two main stages of the livestock supply chain where losses occur are feed production and animal production. To decrease N losses to the environment, measures must address these stages regardless of their geographical location. This is important to avoid the shift of the burden from one production stage to another. For instance, decreasing N losses in one stage could potentially increase losses in successive stages if no measures are implemented. It is important to understand where N losses occur along the chain and determine N loss hotspots. In this section, N flows and opportunities for sustainable N management are described on national, regional and global scales.

### 3.5.1 Global nitrogen flows and losses

In 2010, feed ingested by global livestock systems contained about 106.9 Tg N (Uwizeye *et al.*, 2020). The production of this feed required about 90 Tg N of new N composed of

TABLE 2
**Effects of feeding low crude protein diets on N output of poultry**

| Type of poultry | Protein level in diet | N-related parameter | Level of reduction in N-related parameter |
|---|---|---|---|
| **Broiler** | 16–20 percent | N output | 49.2–65.6 percent |
| | | N output intensity | 12.5–45.8 percent |
| **Broiler** | 20–22 percent with methionine + lysine | N output | 16–38 percent |
| | | N output intensity | 18.8–40.6 percent |
| **Broiler** | 20 percent + enzymes supplementation | N output | 25.8–35.1 percent |
| | | N output intensity | 37.5–43.8 percent |
| **Laying hens** | 11.5–17.5 percent | N output | 26.6–36.3 percent |
| | | N output intensity | 20.0–33.3 percent |
| **Laying hens** | 13.5 percent + enzymes supplementation | N output | Similar |
| | | N output intensity | 12.5–43.7 percent |

*Source:* Adapted from **Malomo, G.A., Bolu, S.A., Madugu, A.S. & Usman, Z.S.** 2018. Nitrogen emissions and mitigation strategies in chicken production. In: B. Yücel and T. Taskin, eds. *Animal Husbandry and Nutrition*. InTechOpen. http://dx.doi.org/10.5772/intechopen.74966

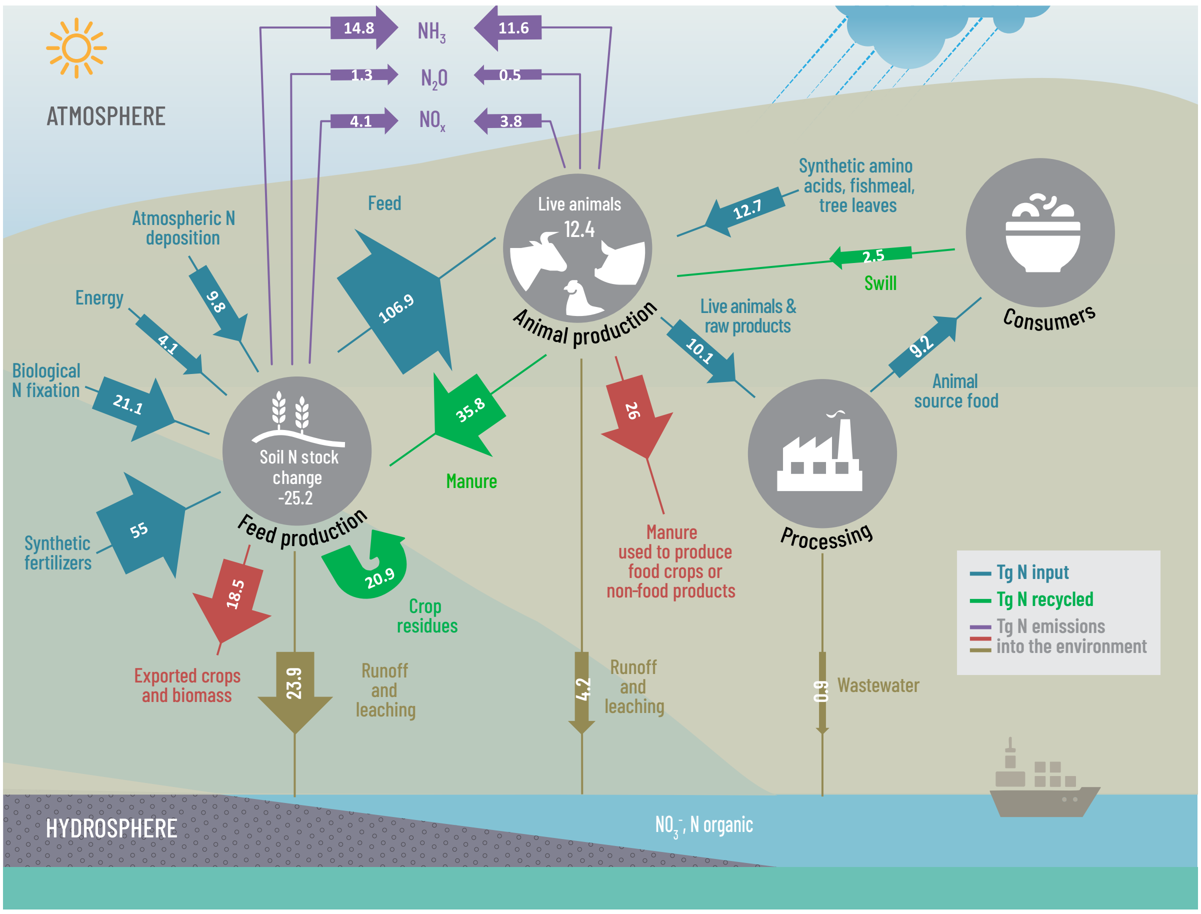
60 percent synthetic N fertilizer, 23 percent biological N fixation, and 11 percent atmospheric N deposition. It relied on 56.7 Tg N of recycled N composed of crop residues (63 percent) and manure (37 percent) used for feed production. It used about 25.2 Tg N from soil N mining mainly in LMICs where access to new N is limited. Furthermore, feed production is associated with about 44 Tg N of N losses to the environment in the form of $NO_3^-$ (54 percent), $NH_3$ (34 percent), $NO_x$ (9 percent), and $N_2O$ (3 percent), as well as 28.1 Tg N lost via runoff and leaching. Figure 14 shows the global N flows of the entire livestock supply chain. Nitrous oxide emissions from feed production exacerbate climate change and are equivalent to 355 Tg $CO_2$eq. Ammonia and $NO_x$ have a high impact on air quality, whereas $NO_3^-$ contributes to water pollution.

In animal production, N inputs were estimated at 122 Tg N/year in 2010 and were sourced from feed (106.9 Tg), synthetic amino acid (12.7 Tg N) and swill (i.e. feed from food waste: 2.5 Tg N). Only 10 Tg N were recovered in final animal products, whereas 28.4 Tg N was lost into the environment in the form of $NH_3$ (41 percent), $N_2$ (29 percent), $NO_3^-$ (15 percent), $NO_x$ (13 percent) and $N_2O$ (2 percent) (Uwizeye *et al.*, 2020). An additional 61.8 Tg N was recovered in manure stored, recycled or deposited on grazing lands. At the processing level, only 0.9 percent of N in final products is lost, mainly through wastewater or organic waste.

Spatial differentiation on a global scale can be seen with several hotspots of emissions. Most emissions take place in South Asia (35 percent), followed by East and Southeastern Asia (28 percent) (Figure 15).

FIGURE 14

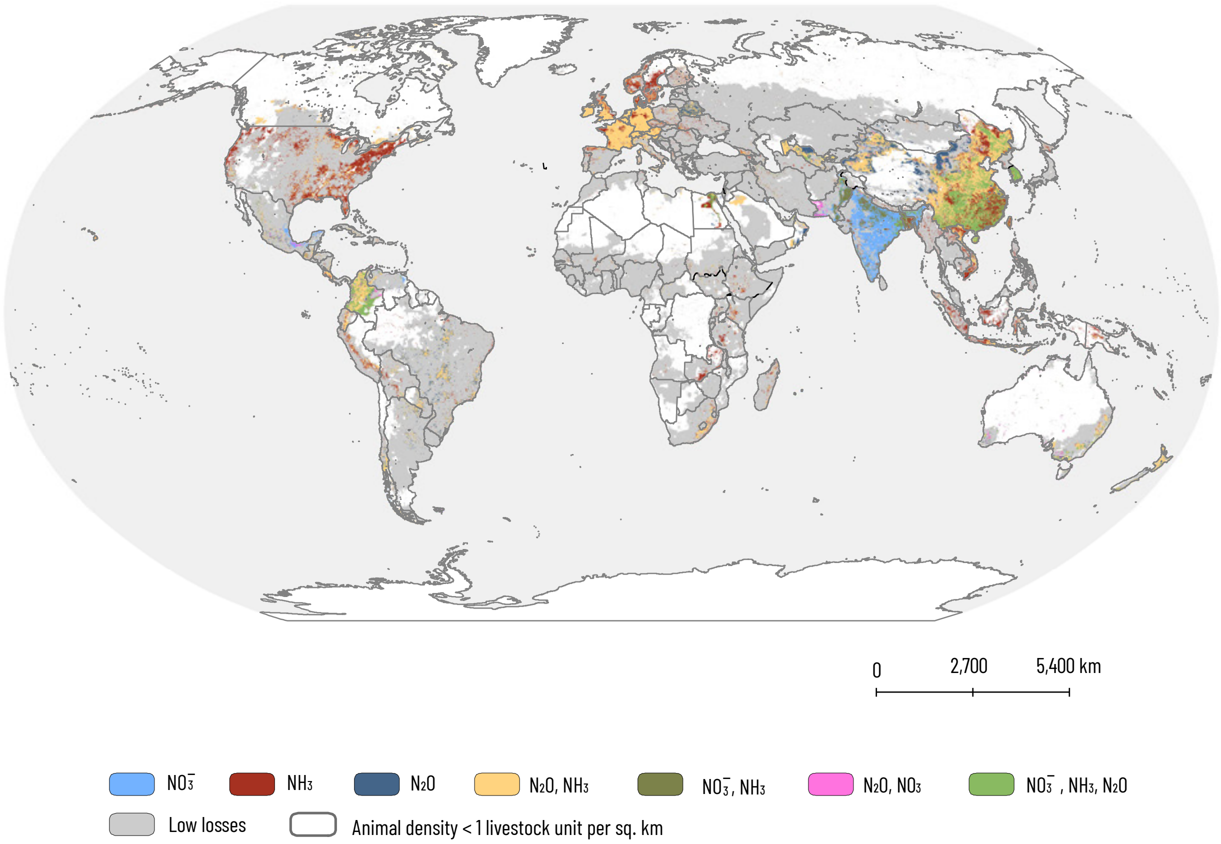
**Global N flows and sources of N emissions allocated to the livestock sector (Tg N/yr)**




*Note:* Nitrogen emissions associated with manure used to produce food crops and non-food products are aggregated.

*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

FIGURE 15
**Hotspots of $N_2O$, $NH_3$ and $NO_3^-$ emissions from livestock supply chains**




*Note:* Refer to the disclaimer on page ii for the names and boundaries used in this map. The map shows classes of hotspots in which one or more N compounds are concentrated. The map was produced using GLEAM 2.0 (https://www.fao.org/gleam/en/). Dotted or dashed lines on maps represent approximate border lines for which there may not yet be full agreement.

*Source:* **United Nations Geospatial.** 2020. *Map of the World.* United Nations. Cited 22 August 2022. www.un.org/geospatial/file/3420/download?token=TUP4yDmF Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

Overall, N emissions along the livestock supply chains are estimated at 65 Tg N, representing circa one-third of total N emissions from anthropogenic activities (Uwizeye *et al.*, 2020). Nitrogen emissions in the form of $NO_3^-$ (28 Tg N) and $NH_3$ (26 Tg N) are dominant, followed by $NO_x$ (8 Tg N) and $N_2O$ (1.8 Tg N). When looking at animal categories, ruminants (cattle, buffalo, sheep and goats) are responsible for most N emissions globally, emitting 45 Tg N/yr, which is 70 percent of the total N emissions from livestock. In all regions, most N emissions come from at least one category of ruminants. Only in Western and Eastern Europe, besides cattle, do monograstrics play a significant role in N emissions. The high concentration of emissions in Southern and Eastern Asia is related to a high density of cattle and buffalo populations. Excreted manure is often poorly managed and application rates of synthetic fertilizer are high, contributing to total N emissions in the region (Beig *et al.*, 2017).

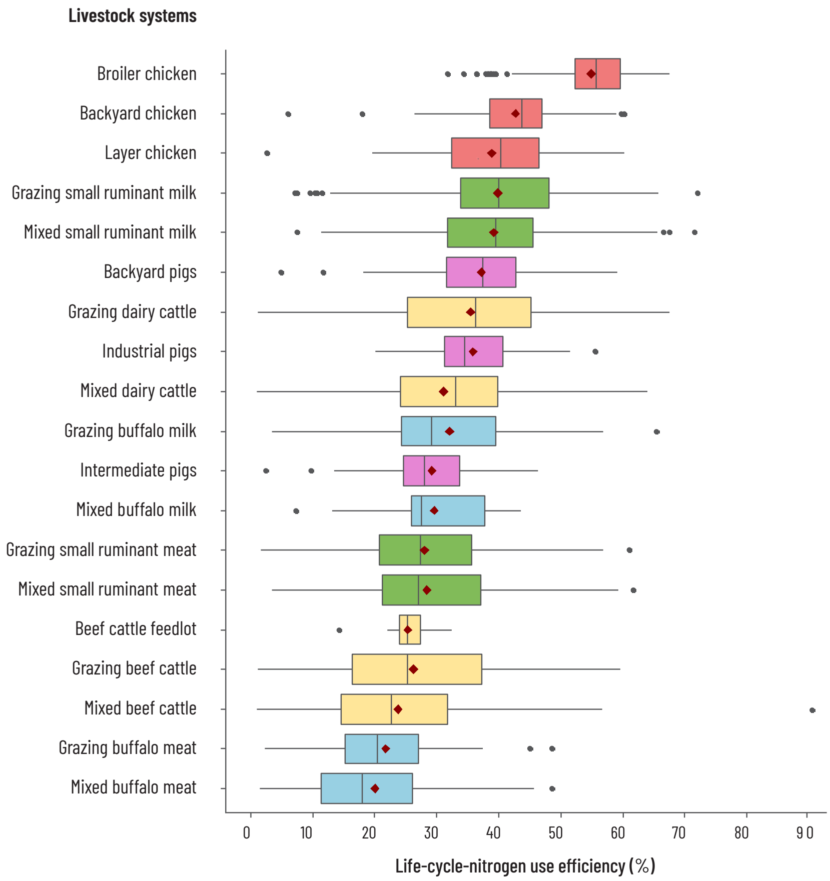
### 3.5.2 Enhancing nitrogen use efficiency in the global livestock supply chain

The life-cycle-NUE indicator was developed to improve the NUE of the entire livestock supply chain (Uwizeye *et al.*, 2020). This indicator refers to the efficiency of recovering N mobilized at each stage of the supply chain into the final animal product, thus the portion of N input that ends up in an animal product for human consumption. A similar approach is described by Sutton *et al.* (2013) as the "full chain NUE", which is defined by the ratio of N in a final product to new nutrient inputs (e.g. N produced through the Haber–Bosch process, biological N fixation, mining of N and $NO_x$ formation). Erisman *et al.* (2018) propose the use of the "whole food chain NUE", which is defined as the N available in food for consumption divided by the newly formed N that was used to produce the food and can be calculated at the country and region level. In this way, it describes the percentage of N invested in food production that is contained in the final product for consumption. All these indicators ultimately indicate how effectively new N mobilized for food production is converted into protein N available for human consumption and, thus, how efficiently N has cycled through the whole production chain. For each step and component of the production chain, efforts can be made to improve NUE, and the combination of measures that improve component efficiencies will contribute to the overall life-cycle-NUE of the supply chain. Through this, objectives to improve efficiency and recycling of N can be set for each stage of the livestock supply chain, while the life-cycle-NUE indicator shows the overall performance of the sector on a global, national or regional scale.

Uwizeye *et al*. (2020) carried out a disaggregated global study on the NUE performance of various livestock supply chains where both newly formed N as well as recycled N (e.g. through manure or crop residues) were included to calculate the life-cycle-NUE. This analysis was conducted for 275 countries and territories for different livestock species and systems, as seen in Figure 16.

Systems with the highest efficiencies are broiler chicken systems and backyard chicken systems for both egg and meat production with life-cycle-NUE values ranging from 32 to 67 percent. Ruminant systems with relatively high life-cycle-NUE (more than 36 percent) include small ruminant systems (both grazing and mixed systems) and dairy cattle in grazing systems. When looking at industrial and intermediate pig systems, median life-cycle-NUE values range from 28 to 36 percent. The lowest efficiencies for N use are found in large ruminant (buffalo, beef and dairy cattle) intensive grazing and mixed systems, as well as feedlot systems. This is in line with previous studies that show the low NUE with which large ruminants convert feed into meat or milk because they need large quantities of feed. There is large variability per country, as

FIGURE 16
**Life-cycle-NUE by livestock species and systems**




*Note:* The coloured boxes delineate the 25 (left-side) and 75 percentile values (right side), the vertical centre lines indicate the median value (50 percentile), diamonds indicate the mean values, the horizontal lines indicate the five (left-end) and 95 percentiles (right-end), and the dots represent outliers. The coloured boxes refer to different livestock types. The smaller the boxes, the smaller the variations in NUE between countries within a certain livestock production system.

*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

can be seen in Figure 16, by the size of the boxes per livestock system, as well as the outliers. When looking at NUE of the whole food system (including both crop and livestock production) for a country or region, Erisman *et al*. (2018) evaluated the different studies carried out. Results showed an average NUE of 5–15 percent, indicating that only a small portion of N invested in global food production is contained in the final products suitable for human consumption. The higher values of life-cycle-NUE found by Uwizeye *et al*. (2020) can be attributed to the fact that they consider N flows, such as recycling of crop residues, manure, and soil N stock change at each stage as output.

The inclusion of livestock in a food production system leads to lower NUE values as it lengthens the nutrient chain and thereby increases the steps and opportunities for N to be lost to the environment (Sutton *et al.*, 2013). It is important to look closely at the livestock supply chain and its different stages to identify opportunities to improve efficiency and resource use and increase the overall sustainability of livestock production. Studies have shown that increasing the efficiency and quality of manure management and recycling increases the overall NUE (Sutton *et al.*, 2013; Uwizeye, 2019) as more products are produced from the same level of new nutrient inputs. For this reason, recycling nutrients and feeding livestock with by-products or swill contributes to increasing overall NUE as well, as less N embedded in feed crops is mobilized to feed livestock.

Focusing solely on NUE does not necessarily lead to a reduction in total N emissions as increased efficiency leads to decreased production costs, which could lead to a consumption surge (Uwizeye, 2019). It is fundamental to combine NUE of livestock systems with control of the sector's expansion, consider other sustainability indicators and address over-consumption of animal products and behavioural change by consumers. For instance, options to shift to a more plant-based diet in high-income countries could increase NUE of agrifood systems, and decrease N waste, pollution, and losses to the environment, while considering trade-offs with other sustainability indicators (Erisman *et al.*, 2018; Leip *et al.*, 2023; Sutton *et al.*, 2013).

### 3.5.3 Embedded nitrogen in international trade

The international trade of food and feed is a key element in the global agrifood system and has increased significantly from the second half of the past century, where circa 25 percent of the food and feed produced globally is traded internationally (Lassaletta *et al.*, 2014b). International trade can contribute to achieving food security and nutrition in countries where domestic production is insufficient and can offer a wider range of food and feed products. The international trade of food commodities plays an important role in the disruption of N cycles and N pollution linked to biodiversity loss, climate change, deforestation, eutrophication and acidification (Billen, Lassaletta and Garnier, 2015; Oita *et al.*, 2016; Wang *et al.*, 2022a). This is specifically the case for international trade linked to the livestock supply chain, with livestock feed being the most important commodity. Approximately 8 percent of total N emissions from the livestock supply chain is embedded in feed and livestock commodities from international trade, which amounts to approximately 5.5 Tg N, of which 1.5 Tg N/yr is entirely attributed to the production of feed, whereas livestock commodities contributed about 4 Tg N/yr (Uwizeye *et al.*, 2020). Figure 17 shows the magnitude of embedded N emissions in the international trade of feed for the major importing and exporting countries.

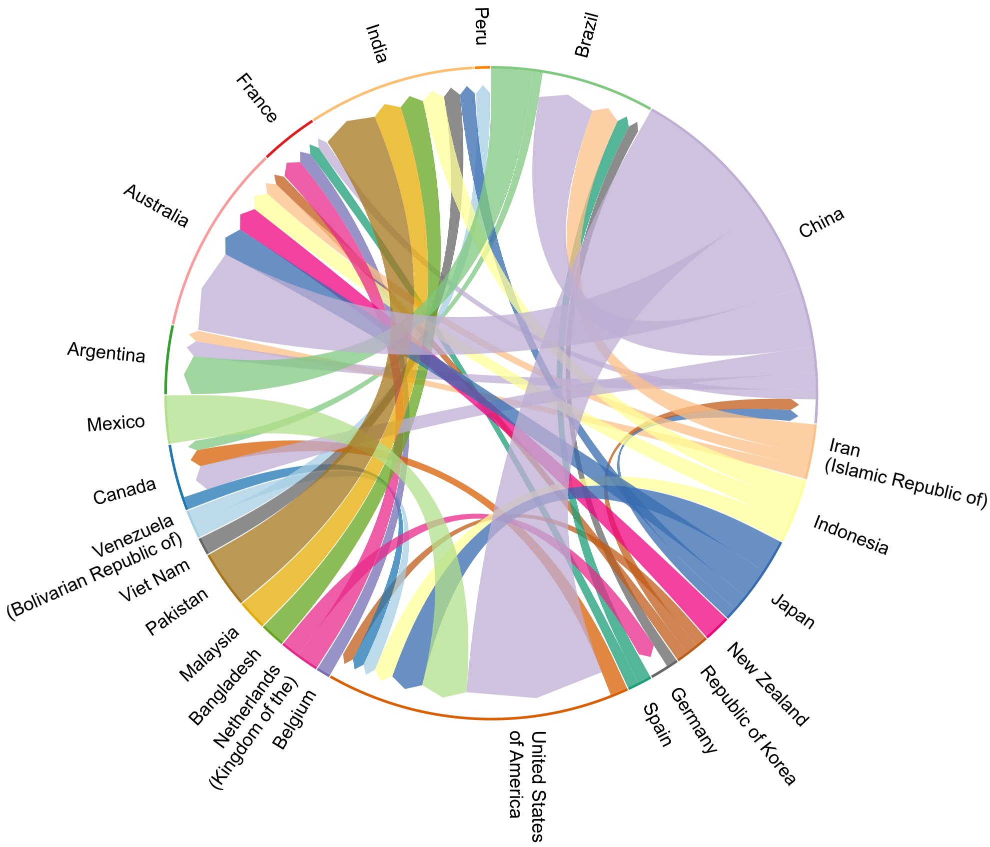
Through international trade, the livestock supply chain is lengthened, which results in an increased number of production steps where N losses can occur. As N losses linked to international trade increase, the life-cycle-NUE typically decreases. Wang *et al*. (2022a) found that for both importing and exporting countries, NUE of the food system decreased between 1963 and 2011 and was associated with an increase in international trade of food and feed, food supply, fertilizer N use and biological N fixation. Furthermore, N balances for both importing and exporting countries were positive and showed an increase over time, concomitant with increased international trade. A positive N balance is associated with an increased amount of N losses to the environment. As international trade plays such a significant role in N transfer within the global agrifood system, associated N losses and NUE, it can play a key role in finding solutions for sustainable N management along the food production chain. Figure 18 shows embedded N emissions in the international trade of feed commodities (cereals and soybean products). These feed products represent some of the major commodities traded in the livestock supply chain, with cereals representing 60 percent of total emissions and soybean products representing 39 percent of total N emissions.

Regarding the livestock commodities, Figure 19 shows N emissions embedded in the dairy and beef cattle products, which include the feed used to produce them. Beef meat represents 31 percent of embedded N emissions in the international trade of livestock products and cattle milk represents 43 percent of total embedded N emissions.

### 3.5.4 Enhancing nitrogen use efficiency at national and regional scale

The options described in section 3.4 to improve NUE have focused on farm-level measurements. This section describes how NUE can be enhanced when looking at the agrifood production system. Integration of crop and livestock systems has been widely accepted as one of the major ways to recycle N (and P) efficiently, decrease systems' reliance on external inputs and minimize losses of N to the environment (Baker *et al.*, 2023; Billen *et al.*, 2021; Castillo *et al.*, 2023; Watson,

FIGURE 17
**Embedded N emissions in international trade of livestock feed**




*Note:* The ribbon colours distinguish the countries of import and export and the arrowhead of each ribbon points to N emissions released in each exporting country by a given importing country. The size of the arrow indicates the relative magnitude of embedded N emissions. The direction of the arrow indicates the attribution of N emissions in imported livestock commodities. For clarity, the figure represents 50 percent of all bilateral trade of feed globally.

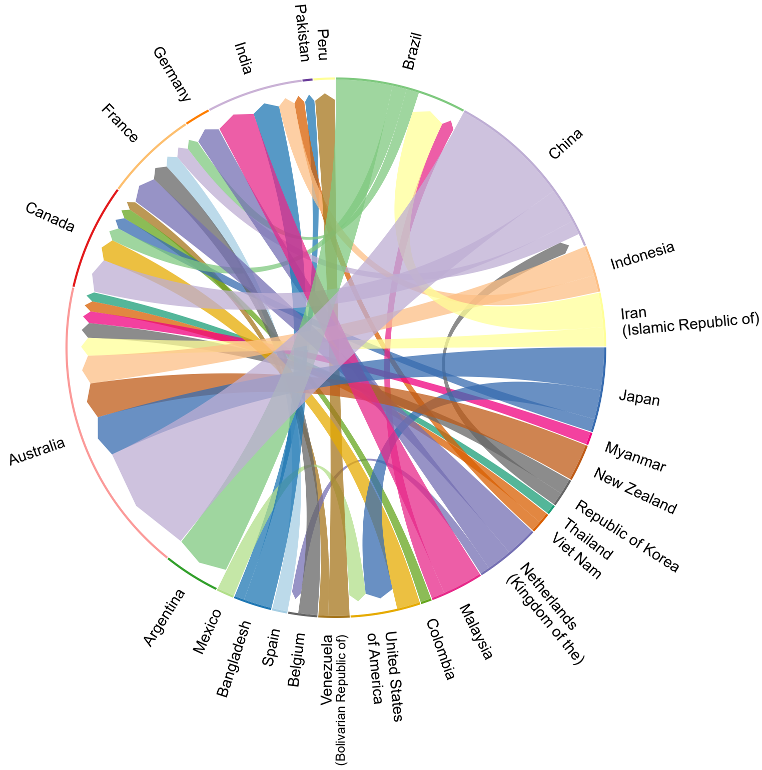
*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

Topp and Ryschawy, 2019; Lassaletta *et al.*, forthcoming). As the development of the livestock sector in different regions (e.g. Northern America, Western Europe and Southeast Asia) has resulted in livestock-dense areas that largely depend on external resources, crop and livestock systems are decoupled, resulting in N hotspots in areas where livestock are kept, and N losses where feed production is prevalent. Ultimately, this results in a disruption of the N cycle and increases the dependence on newly produced N. To create a more balanced system where N recovery is maximized, re-integration of crop and livestock production can be done at farm, national or regional level (Schut *et al.*, 2021). Integrated crop–livestock systems can not only significantly reduce N losses, but they have proven, in different regions and management systems, to increase crop yields and animal productivity compared to specialized systems without crop–livestock integration (Farias *et al.*, 2020; Schut *et al.*, 2021; Sekaran *et al.*, 2021). For instance, Farias *et al*. (2020) and de Faccio Carvalho *et al*. (2018) found soybean systems integrated with livestock generated a 58 and 60 percent increase in soybean yield equivalents, respectively. De Faccio Carvalho *et al*. (2018) found that introducing grazing cover crops in grain production systems increased yields of the grain crops from around 3.5 to 11 percent compared to conventional grain crop production systems, along with additional output from the livestock component.

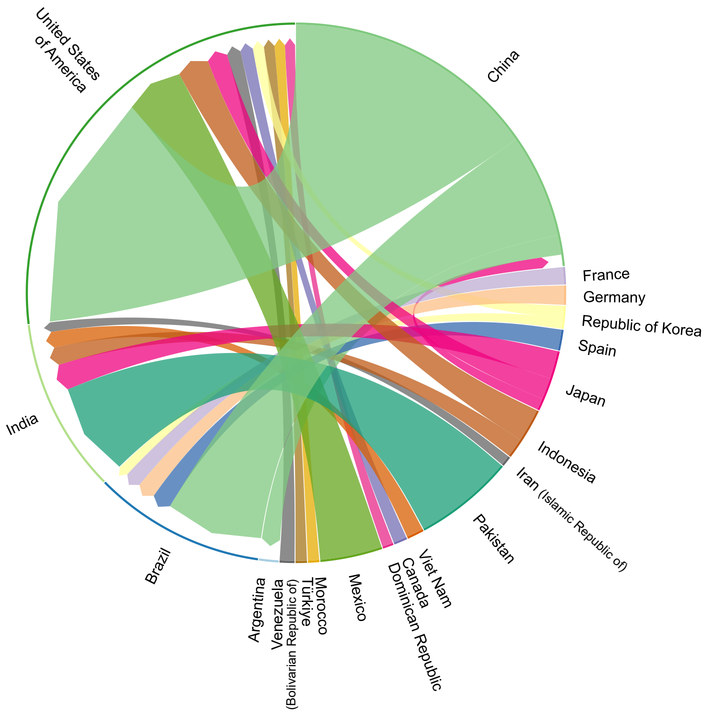
On what level (e.g. farm, national, regional) integration of crop and livestock systems is possible and profitable, both on environmental and economic levels, highly depends on the farming systems and characteristics of the region. For example, on farm level, mixed crop–livestock systems require different skills and activities and might not be favourable for large, specialized farms. Smallholder farms under tropical and sub-tropical conditions have

FIGURE 18
**Embedded N emissions in international trade of cereals (maize, wheat and barley [a]) and soybean products (b)**

**(a)**




**(b)**

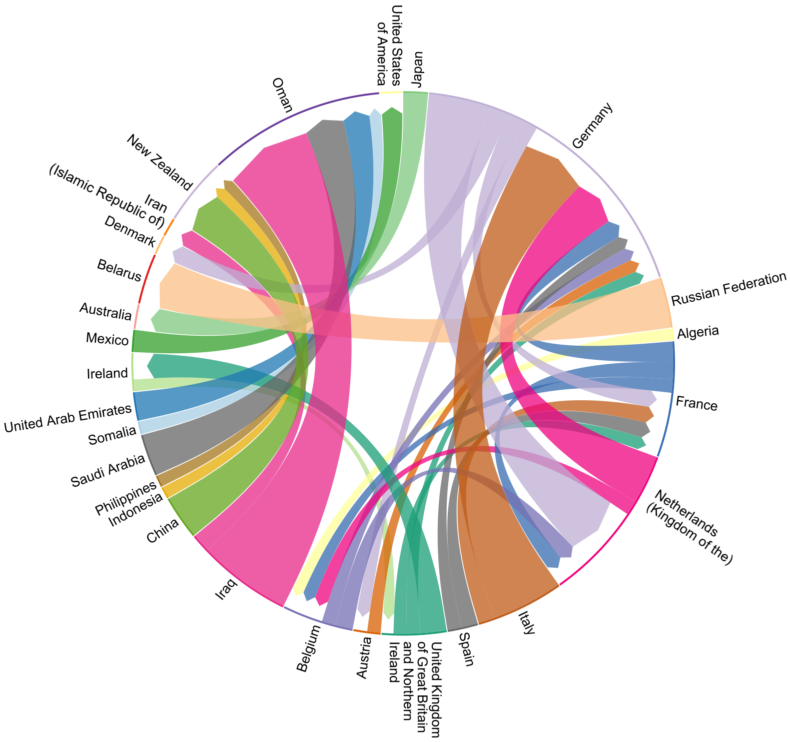



*Note:* The ribbon colours distinguish the countries of import and export and the arrowhead of each ribbon points to N emissions caused in each exporting country by a given importing country. The size of the arrow indicates the relative magnitude of embedded N emissions. The direction of the arrow indicates the attribution of N emissions in imported commodities. For clarity, only major trade flows are represented; the figure represents 50 percent of bilateral trade flows of maize, wheat and barley (a) and 60 percent for soybean products (b).

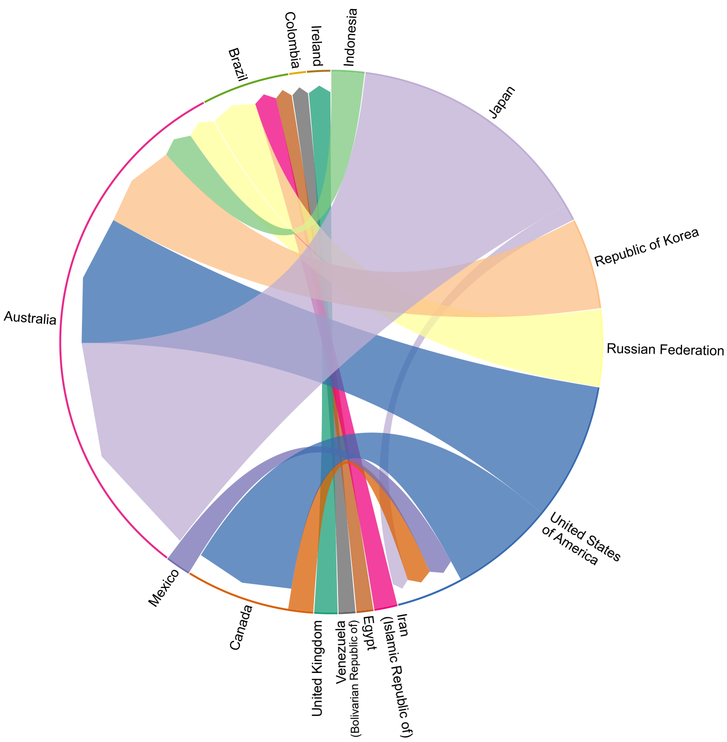
*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F.** ***et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

FIGURE 19
**Embedded N emissions in international trade of dairy products (a) and beef meat (b)**

(a)




(b)




*Note:* The ribbon colours distinguish the countries of import and export and the arrowhead of each ribbon points to N emissions caused in each exporting country by a given importing country. The size of the arrow indicates the relative magnitude of embedded N emissions. The direction of the arrow indicates the attribution of N emissions in imported commodities. For clarity, only major trade flows are represented; the figure represents 40 percent of bilateral trade flows for dairy products (a) and 60 percent for beef meat products (b).

*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

benefited from differentiation of production on the farm, which improves productivity (through improved fertilization of arable land), increases animal health and productivity (through increased availability of feed) and enhances overall resilience of the farming system (Devkota *et al.*, 2022; Rufino *et al.*, 2007; Sekaran *et al.*, 2021). Furthermore, integration of crop and livestock can increase food availability and security for smallholder farmers (Paramesh *et al.*, 2020; Sekaran *et al.*, 2021). For specialized farms, characterized by high technological input and benefits from economies of scale, integration of crop and livestock production is more beneficial on a regional scale (Russelle, Entz and Franzluebbers, 2007; Schut *et al.*, 2021). Between farms, resources such as manure, fodder crops and crop residues are shared, which results in a more efficient use of (land) resources and better utilization of labour throughout the year. Similarly, agreements between shepherds and arable farmers to allow temporary grazing can decrease crop weeds, and increase soil nutrients and access to fresh pastures (Schut *et al.*, 2021). If livestock can be kept outdoors in winter through this kind of exchange, an additional reduction of feed imports can be realized as well (Boyle *et al.*, 2008). Different studies have shown that integrated crop–livestock systems are more sustainable and resilient to external shocks, such as market conditions, fluctuating prices and weather extremes (Bell and Moore, 2012; Moraine, Duru and Therond, 2017; Schiere, Ibrahim and van Keulen, 2002). Schut *et al*. (2021) outlined the different ways in which specialized farms can collaborate as: (1) exchange products and resources while keeping the same crop rotations as before; (2) adapt crop rotations to complement the needs of the livestock farmer (i.e. produce feed); and (3) increase mutual benefits and synergies by exchanging fields that allow increased production of high-value crops or widen the crop rotation. Adoption of any of these three options for specialized farms is dependent on factors such as distance between farms, preferred level of independence by farmers and other socio-cultural barriers. All three options can be of significant value for the development of a sustainable agrifood system if, through this, nutrient cycles are closed, resources are used more efficiently, losses of N to the environment are minimized and soil quality through organic fertilization is improved. Additionally, integrated crop–livestock systems on a regional scale can enhance biodiversity and contribute to ecosystem services (Baker *et al.*, 2023; Billen *et al.*, 2021; Schut *et al.*, 2021).

Overall, the adoption of integrated crop–livestock systems on a regional scale, optimizing collaboration between specialized farms, can result in higher efficiency of resource use, and a reduction of external inputs with similar or higher yields. Using manure, significant amounts of synthetic fertilizers can be replaced, resulting in improved N balances at a national level. Furthermore, the NUE of the agrifood system can be improved, with research showing that low-efficiency components, which is typical for livestock systems, can significantly increase the NUE of the crop system, resulting in a high NUE of the overall integrated crop–livestock systems (Castillo *et al.*, 2023). Lastly, integrated crop–livestock systems can greatly contribute to the productivity, resilience, livelihoods and food security of smallholder farmers.

## 3.6 CASE STUDIES

### 3.6.1 Using swill as feed

In a circular food system, one of the primary focuses is to avoid waste from agricultural production processes to maintain nutrients within the system (Chapter 5). There will be unavoidable waste streams that are linked to harvesting and food processing steps (i.e. crop by-products and waste streams), as well as food losses at the consumer stage (typically known as swill), which encompass total FLW streams from the agrifood system. These can potentially form a valuable source of livestock feed, especially for monogastric animals such as pigs and poultry. By-products from the processing industry are already being used as feed (e.g. vegetable, baking, brewing and sugar beet by-products). This encompasses only a small part of feed used in pig and poultry production, as grains and oil seed cakes make up most of the diet in industrial systems (Mottet *et al.*, 2017). The amount of FLW potentially available for feed is significantly higher. Currently, swill is used as feed in several countries, including Japan, Republic of Korea and Thailand. These countries have successfully implemented a recycling system wherein swill is collected and treated, ensuring that the quality and safety of the feed is maintained (zu Ermgassen *et al.*, 2016). On the other hand, there are still many countries where the use of swill for feed is prohibited due to concerns about public health and infectious disease transmission. Currently, in the whole of the European Union and the Kingdom of Great Britain and Northern Ireland, swill feeding has been prohibited altogether after the foot-and-mouth disease outbreak in 2001, which was caused by illegal (untreated) swill feeding to ruminants (zu Ermgassen *et al.*, 2016).

In a circular food system, pigs and poultry are used to convert waste streams into high-quality nutrients for human consumption. Including swill from the FLW stream to feed monogastric livestock would contribute to the development of a circular agrifood system and could have multiple benefits, such as decreasing losses of N from the agrifood processing chain, replacing high-quality livestock feed (i.e. cereals and soybeans), and thereby decreasing land use for livestock production (Boumans *et al.*, 2022). Furthermore, Uwizeye *et al*. (2019) showed that the replacement of conventional feed with swill increases the life-cycle-NUE of the livestock production chain by 6–30 percent, depending on the ratio of swill in the feed and the original proportion of grains and soybeans. With the right treatment techniques, incentives to include FLW in livestock feed, and certificates for animal products fed with swill, Japan and Republic of Korea have successfully recycled around 36 percent and 43 percent of household waste, respectively (zu Ermgassen *et al.*, 2016). As the implementation of swill feeding has been proven to be successful in these countries, several studies have outlined the possibilities of implementing a similar system in regions where this is still illegal (Boumans *et al.*, 2022; zu Ermgassen *et al.*, 2016). Furthermore, both studies show that swill feeding has no negative effects on the growth of the animal and the quality of the end product. Recycling of FLW for livestock feed is seen as a robust and efficient measure to reduce N losses, increase NUE of the livestock production chain and to play a role in the development of a circular agrifood system.

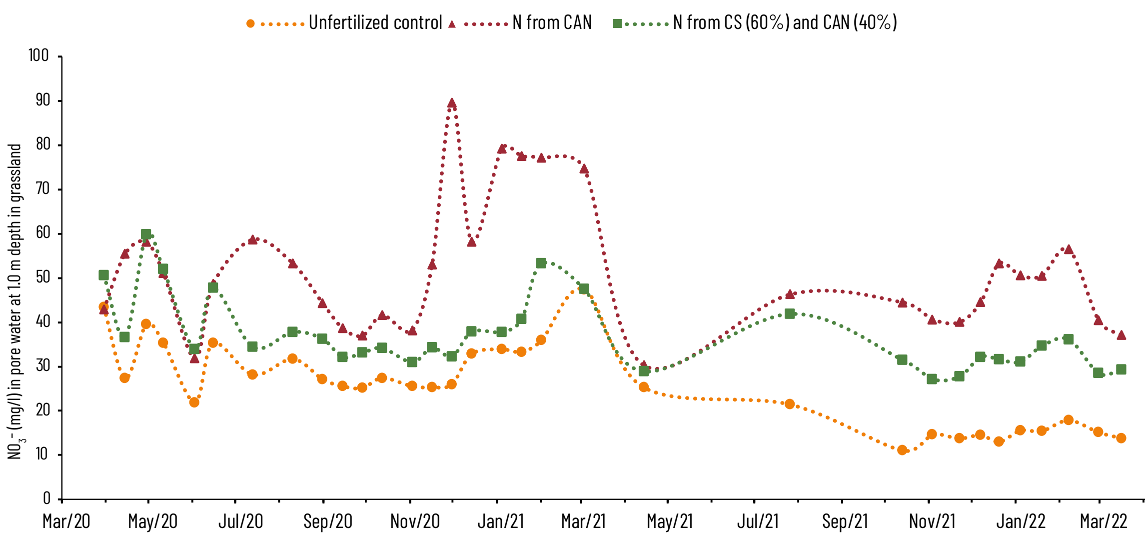
### 3.6.2 Decrease nitrate leaching through the application of cattle slurry

The leaching of $NO_3^-$ from agricultural soils is one of the main sources of N losses from agriculture. Through this, $NO_3^-$ end up in ground- and surface water where it affects (drinking) water quality and can result in eutrophication when $NO_3^-$ concentrations are high. Reduction of $NO_3^-$ leaching plays a key role in preserving water quality and wetlands while it increases NUE in the agricultural production cycle. The case study presented here is based on results from a field experiment by de Boer *et al*. (2024), where research done on the level of $NO_3^-$ leaching from grasslands fertilized with cattle slurry and synthetic fertilizer was compared to the use of synthetic fertilizer only. This is especially applicable to dairy farms in temperate regions with permanent grasslands. In these systems, where cattle are either housed year-round or grazed during the spring/summer season, cattle slurry is collected and can be used as fertilizer for grasslands and croplands. Through this, N excreted through manure can be recycled back into the system.

Through a two-year field experiment, applications of either 100 percent synthetic fertilizer calcium ammonium nitrate (CAN) or a combination of 40 percent CAN and 60 percent cattle slurry (CS) were done on cut grassland on a leaching-sensitive sandy soil. Applications were made during five harvest cycles per growing season. The results showed that the $NO_3^-$ concentration in pore water was 44 percent lower for the CS-CAN compared to CAN only in the first growing season, and 35 percent lower for the CS-CAN combination in the second growing season, while herbage N uptake was similar for both treatments (Figure 20).

The N present in cattle slurry can only be lost through leaching after mineralization and nitrification of the organically bound N. This slow process, combined with a large plant uptake potential of the organic N, suggests a lower risk of N losses. The study by de Boer *et al*. (2024) confirms that the use of organic fertilizer through cattle slurry

FIGURE 20
**Nitrate concentration (mg/l) in pore water at 1.0 m below field level**




*Note:* CAN refers to calcium ammonium nitrate synthetic fertilizer, CS refers to cattle slurry.

*Source:* Adapted from **de Boer, H.C., van Mullekom, M. & Smolders, A.J.P.** 2024. Lower nitrate leaching from dairy cattle slurry compared to synthetic fertilizer calcium ammonium nitrate applied to grassland. *Environmental Pollution*, 344: 123088. https://doi.org/10.1016/j.envpol.2023.123088.

can substantially reduce $NO_3^-$ leaching into groundwater. This not only benefits water quality but decreases the use of synthetic fertilizers and increases N recycling and NUE. Cattle slurry forms a valuable resource that can increase N recycling and avoid substantial N losses when applied on time.

### 3.6.3. Assessment of nitrogen flows and use efficiency in dairy system flows in East Africa

Dairy systems in countries such as Kenya, Rwanda, Uganda and the United Republic of Tanzania contribute to high-nutritional dairy products and support rural livelihoods. Most milk production occurs in traditional and agro-pastoral systems, except in Rwanda and Kenya, where improved crossbreed or exotic dairy cattle dominate. Milk productivity remains low due to constraints related to animal health, water availability, and feed resources. The Global Livestock Environmental Assessment Model (GLEAM), based on Uwizeye *et al.* (2020) methodology, estimates N flows, efficiency and emissions from different dairy systems. Kenya's dairy cattle population of 4.9 million comprises indigenous (54 percent) and exotic (46 percent) breeds (Kenya National Bureau of Statistics, 2019). Rwanda emphasizes sustainable livestock management, with 1.6 million cattle, including local, crossbreed and exotic breeds (NSIR, 2020). In the United Republic of Tanzania, traditional dairy farming relies on Tanzania Shorthorn Zebu, while intensive systems combine exotic breeds with local cattle (The United Republic of Tanzania, 2021). Uganda's dairy production includes both traditional and commercial systems, with small-scale household production and limited decision-making power for women and youth (FAO and NZAGRC, 2019).

In East Africa, N intake by dairy cattle varies from 25 kg N/animal/yr in Uganda to 46 kg N/animal/yr in Kenya and Rwanda. The excretion of N in manure ranges from 22 kg N/animal/yr in Uganda to 41 kg N/animal/yr in Rwanda and Kenya (Figure 21). Approximately 18–28 percent of the excreted N is lost into the environment in the forms of $NH_3$, $NO_x$, $N_2O$ and $NO_3^-$ (Figure 22). Ammonia emissions dominate, accounting for about 60 percent of N emissions in Rwanda and Kenya, and 39–41 percent in Uganda and the United Republic of Tanzania. These differences are linked to the prevalence of zero-grazing or intensive systems in Rwanda and Kenya.

Nitrous oxide emissions are higher in the United Republic of Tanzania (5 kt N/yr) compared to Kenya (0.9 kt N/yr), Rwanda (0.6 kt N/yr) and Uganda (0.58 kt N/yr). Similarly, $NO_3^-$ emissions are elevated in the United Republic of Tanzania (9 kt N/yr) relative to other countries. Nitrogen oxides emissions, resulting from manure used as a fuel source, are significant in the United Republic of Tanzania (80 kt N/yr) and Uganda (29 kt N/yr). The details of N flows and emissions are provided in Table 3.

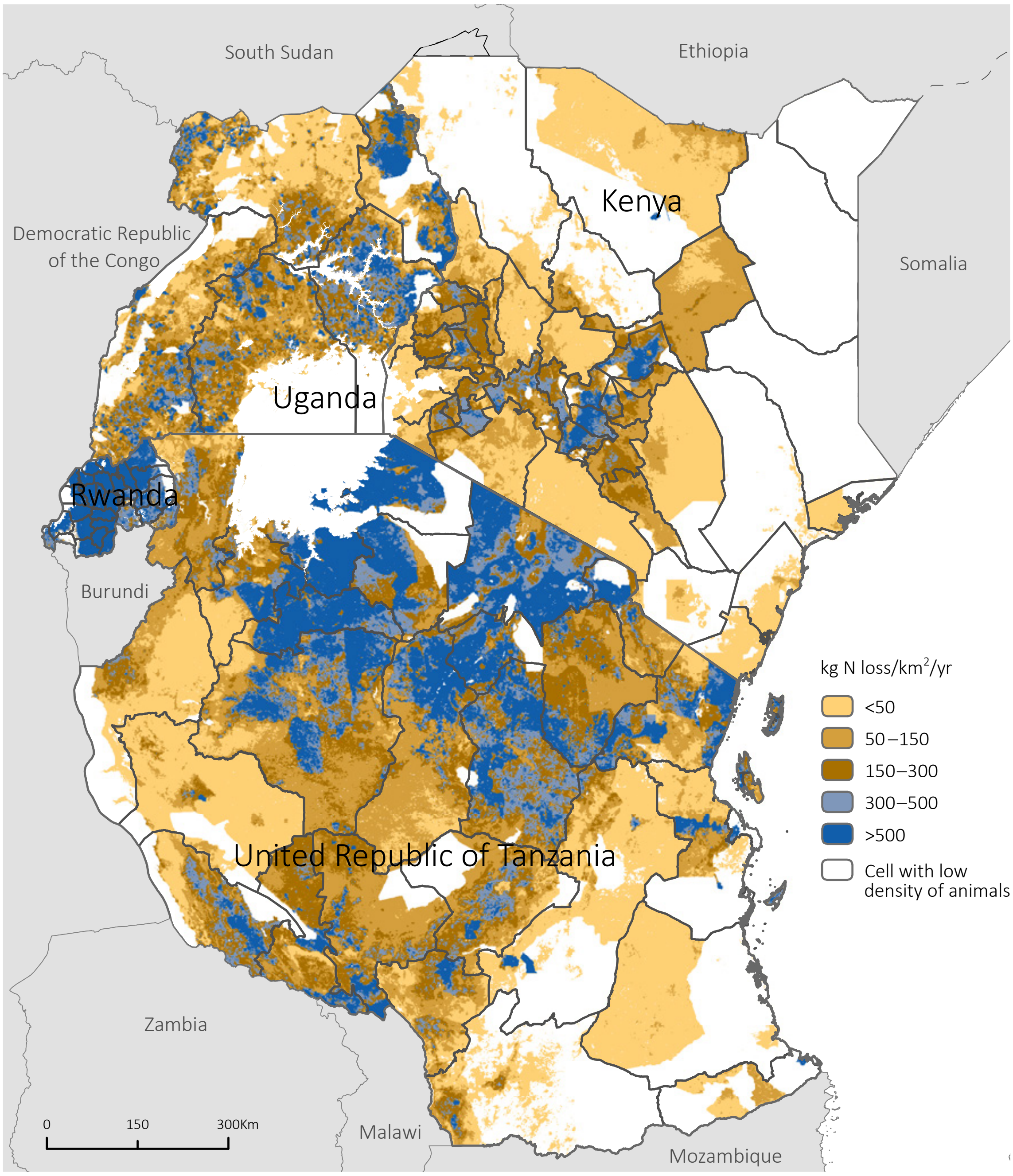
TABLE 3
**Detailed N flows and emissions from dairy systems in East Africa**

| | Population | N intake | N excreted | N losses | N recycled | $NH_3$ | $N_2O$ | $NO_x$ | $NO_3^-$ leaching and runoff |
|---|---|---|---|---|---|---|---|---|---|
| | Million head | Thousand tonnes of N | | | | | | | |
| **Kenya** | 4.9 | 169.3 | 150.9 | 32.1 | 118.8 | 19.3 | 0.9 | 0.3 | 1.4 |
| **Rwanda** | 1.8 | 81.8 | 73.4 | 20.3 | 53.1 | 12.4 | 0.6 | 0.2 | 0.7 |
| **United Republic of Tanzania** | 37.3 | 1094 | 995.4 | 248.6 | 746.8 | 96.0 | 5.2 | 80.2 | 9.1 |
| **Uganda** | 24.2 | 370.8 | 317.4 | 56.7 | 260.7 | 23.4 | 0.6 | 28.7 | 0.4 |

*Source:* Authors' elaboration.

FIGURE 21

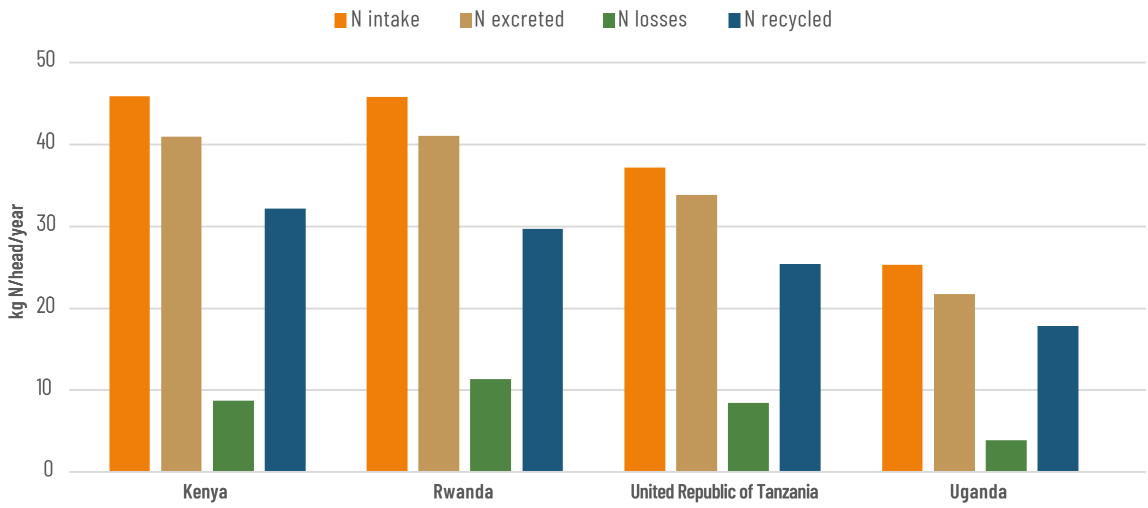
**Total N losses from manure management systems for dairy cattle systems in East Africa (kg N loss/km²/yr)**




*Note:* Detailed results are available on the GLEAM dashboard (https://www.fao.org/gleam/dashboard/en/).
Refer to the disclaimer on page ii for the names and boundaries used in this map.

*Source:* **United Nations Geospatial.** 2020. *Map of the World.* United Nations. Cited 22 August 2022. Modified by authors based on GLEAM 3.0. www.un.org/geospatial/file/3420/download?token=TUP4yDmF

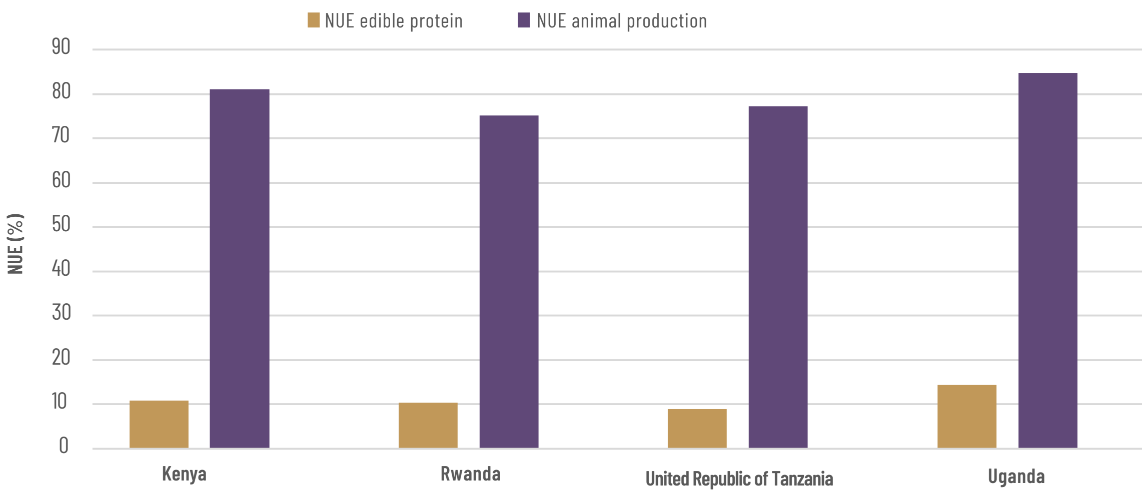
FIGURE 22
**Nitrogen flows in dairy systems in East Africa based on GLEAM 3.0 (kg N/head/yr)**




*Source:* Authors' elaboration based on GLEAM 3.0.

Nitrogen use efficiency, which measures the recovery of N inputs into final edible livestock products (milk and meat), varies from 9 percent in the United Republic of Tanzania to 14 percent in Uganda. When considering manure recycling for crop production, NUE increases from 75 percent in Rwanda to 85 percent in Uganda (Figure 23). These results reflect the economic value and role of manure as the main source of organic nutrients to crops, which contributes to the circular bioeconomy of agrifood systems in East Africa (Leip *et al.*, 2019).

FIGURE 23
**Average on-farm NUE of dairy systems in East Africa**




*Note:* On-farm NUE is given for NUE of edible proteins and NUE of animal production.
*Source:* Authors' elaboration based on GLEAM 3.0.

## 3.7 CONCLUSION AND KEY MESSAGES

The livestock supply chain plays a significant role in global and regional disruption of N cycles and N losses to the environment. Decoupling of livestock and local feed production has resulted in areas of feed production concentrated in specific regions and livestock-dense areas depending on concentrated feed from outside their region. These long supply chains include multiple steps where N is lost to the environment, resulting in even lower NUEs for livestock systems compared to crop systems.

The major source of N losses of livestock systems is feed production associated with high synthetic fertilizer use and land-use change. Emissions of N from manure accumulation in livestock-dense areas play an important role as well. Through this, livestock production is responsible for about one-third of total anthropogenic N emissions.

For sustainable management of N, improving NUE in livestock systems is key, and many improvement pathways are available, depending on region and livestock production system. On-farm interventions can be focused on best management practices for manure and fertilizer application in crop and grassland systems, including low-emissions spreading techniques for manure and improved feeding strategies such as low-protein feeding and feeding of waste streams. Improvement options for housing systems and manure storage and processing techniques can decrease $NH_3$ emissions and leaching of $NO_3^-$. Lastly, improved grassland management and increased grazing time for ruminant production systems can enhance NUE and decrease N losses.

Beyond on-farm measures, integration of crop and livestock systems on a regional level can improve N cycling, reduce N losses, and enhance ecosystem services and biodiversity. Re-integration of livestock and crop production focusing on decreasing the reliance on external inputs and maximizing N recovery in the system can substantially increase the NUE of the agrifood system.

Chapter 4

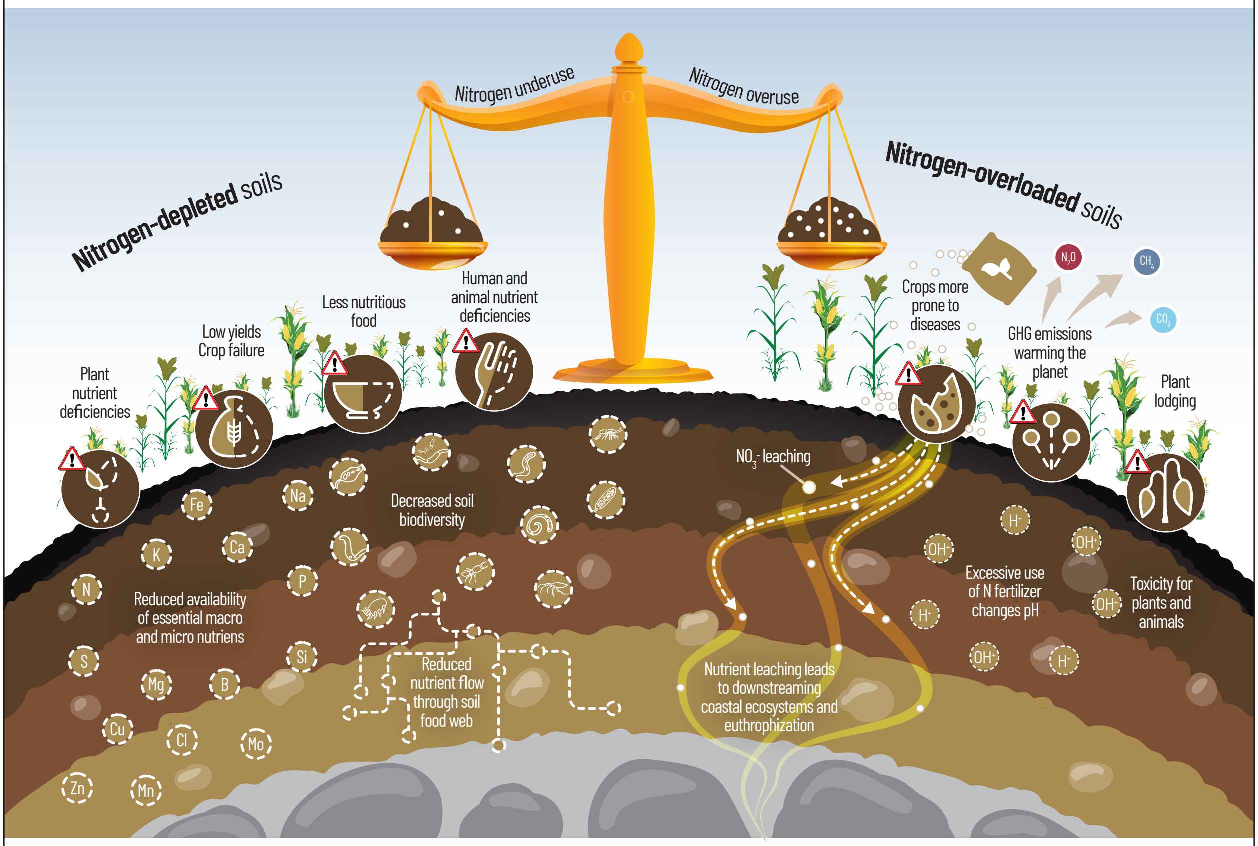
# Impacts of nitrogen losses on climate and ecosystems

## 4.1 INTRODUCTION

Anthropogenic emissions of N into ecosystems are at the centre of debates on various prevalent issues. Notably, N addition is essential for crop growth and production, but its excessive use in agriculture has led to negative impacts on soil, water, air, climate, biodiversity, and human health. Some major adverse effects include increased emissions of GHG, harmful algal blooms, hypoxia of fresh and coastal waters, N deposition onto forests and other natural areas, and crop exposure to elevated ozone ($O_3$) levels, which reduces crop yields, root growth and photosynthetic rates (Nowroz *et al.*, 2024). Conversely, a shortage of N undermines agricultural productivity, contributes to land degradation, affects the cycling of other soil nutrients, and diminishes soil quality (Figure 24) (Vitousek *et al.*, 2010).

The unwanted consequences of N can be further aggravated by climate change and vice versa. Because climate alters N dynamics, variation in climatic drivers brought by climate change is likely to enhance the weakening of ecosystems and alter their responses to N. Moreover, the induced effects on ecosystems can exacerbate climate change, creating a positive feedback loop. For example, both excess N and warming can stimulate soil microbial activity in natural ecosystems, increasing the potential for carbon dioxide ($CO_2$)

FIGURE 24
**The challenge of soil N imbalance**




*Source:* Authors' elaboration based on **FAO.** 2022. *Soils for nutrition: State of the art*. Rome, FAO. https://doi.org/10.4060/cc0900en and methane ($CH_4$) emissions. This added GHG to the atmosphere can stimulate more warming.

Assessing and understanding the impacts of N on ecosystems is critical. Once in the environment, N cycles through various oxidized and reduced forms via biological and chemical processes, allowing a single emitted N molecule to initiate a series of effects – both positive and negative – known as the N cascade (Galloway *et al.*, 2003). These effects can vary widely depending on ecosystem type and its resilience. Moreover, N impacts vary in time and space due to factors such as changes in land use, agricultural management, and weather patterns.

Despite these challenges, N assessments have been conducted in different regions of the world, including California, Pakistan, India and Europe. These assessments generated valuable insights into the known and unknown of the N challenge and paved the way for the formulation of best practices and solutions in these areas. Nevertheless, these assessments tend to be general and do not adequately address N losses under various soil, crop, water management, and climatic conditions and scenarios. These assessments have focused on managing surplus N and have not been conducted in areas where N-depleted soils are a concern. Many regions around the world face inadequate access to N, leading to food shortages and soil nutrient deficiencies. It is important to acknowledge that these areas can face growing instances of N depletion, especially when exporting commodity crops to other countries, which can lead to N-related environmental challenges.

In this chapter, the effects of N losses on terrestrial and aquatic ecosystems in a changing climate are described, including the reciprocal feedback between climate change and N (sections 4.2 and 4.3). Sections 4.4 and 4.5 describe the N processes affecting water, air and soil, such as eutrophication, acidification and alteration of GHG, which in turn affect biodiversity, ecosystem processes and human health. Section 4.6 reviews current approaches and advancements in assessing the impacts of N on ecosystems.

## 4.2 NITROGEN IN TERRESTRIAL AND AQUATIC ECOSYSTEMS

The largest reservoir of N in the biosphere is $N_2$ gas,which makes up 79 percent of dry air and is wholly unavailable to most organisms. Dinitrogen gas fixing organisms transform $N_2$ into organic N compounds available for most organisms for the synthesis of amino acids and other metabolic products. Organic N can be further mineralized to $NH_4^+$, which can be taken up by plants, volatilized as $NH_3$, or immobilized into microbial biomass. If not, $NH_4^+$ is converted to nitrite ($NO_2^-$) and then to $NO_3^-$ by the process of nitrification, which includes an intermediate step producing $N_2O$. Nitrate can then be (1) taken up by plants or microbes, (2) leached out of soils, or (3) denitrified. Denitrification cycles N into different forms, including $NO_x$ and $N_2O$, before returning N to the atmosphere as $N_2$. Nitrogen cycles among plants, animals, microorganisms, soils, solutions, sediments, and between terrestrial, aquatic and atmospheric environments.

Before human alteration, two natural processes transfer N from $N_2$ to biologically available forms – lightning and biological N fixation. Because the global agrifood system heavily relies on the addition of N to agriculture, several N-fixing pathways have been added to the natural N cycle, including the industrial fixation of $N_2$ for use as fertilizer, the cultivation of crops with the capacity to fix N symbiotically and the application of manure and organic fertilizers. About 51 Tg N/yr have been added globally from the use of synthetic fertilizers (Uwizeye *et al.*, 2020). As a result, human activities have substantially enhanced N fixation in terrestrial ecosystems (Vitousek *et al.*, 1997), further enhancing the processes of nitrification, denitrification and N leaching. The projected 50 percent increase in global synthetic fertilizer use by 2050, in comparison to the levels recorded in 2012 (FAO, 2018b), is expected to lead to a significant upsurge in $N_2O$ emissions from agricultural soils. This situation presents a substantial obstacle to achieving the climate objectives of the Paris Agreement, which seeks to restrict global warming to 1.5 °C or well below 2 °C above pre-industrial temperatures. Linked with increasing global fertilizer N use is the $NO_3^-$ pollution of ground and surface waters. Globally, 60 percent of areas with elevated $NO_3^-$ in groundwater occur in croplands (Shukla and Saxena, 2018). Liu and Greaver (2010) reported that out of the annual total N outflow from croplands estimated at 148 Tg N/yr in 2010, N leaching accounts for 23 Tg N/yr, and soil erosion for 24 Tg N/yr.

As N input to terrestrial ecosystems increases, N availability eventually exceeds microbial and plant demands, leading the ecosystem to N saturation. This phenomenon was reported in many terrestrial ecosystems including forests (Yu *et al.*, 2018), croplands (Tian *et al.*, 2020), and grasslands (Bai *et al.*, 2010). Prolonged N loadings can saturate the capacity of terrestrial ecosystems to store N, diminish soil N retention (Niu *et al.*, 2016), and increase N leakage to the atmosphere and waters. Ecosystem N saturation likely occurs when the maximum plant photosynthetic rate is achieved or because of other limiting factors, such as light, water and other nutrients (Hautier, Niklaus and Hector, 2009; Peng *et al.*, 2017, 2019). Another reason is it impedes microbial N processing (Niu *et al.*, 2016), which is a prerequisite for the formation of sequestrable N. This impediment is due to imbalances of N over C inputs, leading to a shortage of C needed for microbial processing. As a result, organic N is converted to $NH_4^+$ and then to $NO_3^-$, which can be subjected to leaching and denitrification processes. Nitrogen that leaves terrestrial ecosystems can enter aquatic ecosystems through streams, subsurface

N runoff, shallow subsurface flow paths, groundwater exchange, N deposition, and fixation (Earl, Valett and Webster, 2006). Once in the water, N dynamics are governed by the same N cycling processes: mineralization, nitrification, denitrification and immobilization.

Climate change has a direct influence on N dynamics through the impacts of heat, rainfall, and other climatic drivers. Conversely, N dynamics reciprocally affect climate change. The subsequent sections explore these relationships.

## 4.3 NITROGEN AND CLIMATE CHANGE

### 4.3.1 Nitrogen dynamics are altered by climate change

Climate exerts substantial effects on N cycling processes and transport within and between aquatic and terrestrial ecosystems through warming temperatures and alterations in temperature and precipitation patterns. For example, the effects of climate change on wet N deposition, primarily caused by changes in precipitation, are much higher than on dry deposition (Zhang *et al.*, 2019b). This results in a shift in the relative contributions of wet and dry deposition to overall N deposition on terrestrial and aquatic ecosystems. Marschner (1995) found that direct plant toxicity due to wet N deposition is relatively uncommon, especially among non-vascular plants. Dry deposition of $NH_3$ can induce plant toxicity.

In terrestrial ecosystems, SOM decomposition is temperature-sensitive (Davidson and Janssens, 2006), and soil warming due to increased air temperature often intensifies N mineralization and nitrification rates through effects on enzymatic activity until a maximum temperature is reached. Moreover, soil moisture availability affects the temperature sensitivity of SOM decomposition (Davidson and Janssens, 2006), controls denitrification rates, regulates the movement of N substrates and enzymes, and influences soil oxygen availability (Butterbach-Bahl *et al.*, 2013). Climate change-induced variations in temporal and spatial dynamics of temperature and precipitation can create the so-called hot moments and hotspots for $N_2O$ emissions. Nitrous oxide emission rates are enhanced in wetter and warmer conditions while they are dampened in drier and warmer conditions. The succession of wet and dry cycles may promote nitrification–denitrification coupled processes or mobilize N to aquatic ecosystems. Nitrogen oxides emissions through microbial processes are enhanced by rising temperatures as well (Romer *et al.*, 2018).

The rates at which N enters aquatic ecosystems are largely influenced by precipitation and changes in precipitation patterns, which can accelerate or decelerate water flow. Changes in precipitation affect N processing and the residence time of waters that are essential for N removal (Howarth *et al.*, 2012). In regions experiencing droughts, reduced flow rates and water levels impede hydrological connectivity in terrestrial landscapes, promoting N retention. This retention is due to the inhibition of both nitrification and denitrification. Conversely, when precipitation follows, there is a surge in nitrification, leading to higher N concentrations in the drained waters and subsequent leaching to the aquatic ecosystems. In contrast, in regions experiencing increased precipitations or intense storms, increased water flow may increase leaching and export of N through terrestrial landscapes and to aquatic ecosystems. In either case, more frequent blooms of harmful algal species are possible.

### 4.3.2 Climate change is altered by nitrogen dynamics

It is well-established that human activities involving N present notable trade-offs in terms of their impact on climate change. Certain activities contribute to warming effects that exacerbate it. Conversely, others generate cooling effects, potentially mitigating or counteracting the overall warming trend. For example, emissions of $N_2O$ from agriculture may generate a strong long-term warming effect, whereas N deposition and fertilization enhance plant growth leading to increased photosynthesis and subsequent $CO_2$ removal from the atmosphere. The same process (N deposition and fertilization) can generate a small warming effect by increasing $CH_4$ efflux from the soil (Liu and Greaver, 2009). Studies that examined these trade-offs indicated modest cooling effects on climate from N activities, indicating that N warming effects outweigh their climate benefits. In this section, the state of knowledge on N-induced cooling and warming on the global climate considering both N direct and indirect effects is explored.

#### *Direct effects*

Nitrogen directly affects climate through the emissions of $N_2O$, a potent GHG with unique characteristics that significantly influence the Earth's energy balance. Nitrous oxide has a long atmospheric lifetime (about 116 years), high global temperature change (GTP) of 233 times that of $CO_2$ for a 100-year timescale, and high global warming potential (GWP) that is 273 times that of $CO_2$ and is set to rise around 50 percent through the 21st century (Forster *et al.*, 2021). While $N_2O$ rarely gets mentioned compared to $CO_2$ and $CH_4$, reducing $N_2O$ emissions is necessary to keep global warming under 1.5 °C by the end of this century, considering its role in depleting the $O_3$ layer in the stratosphere.

#### *Indirect effects*

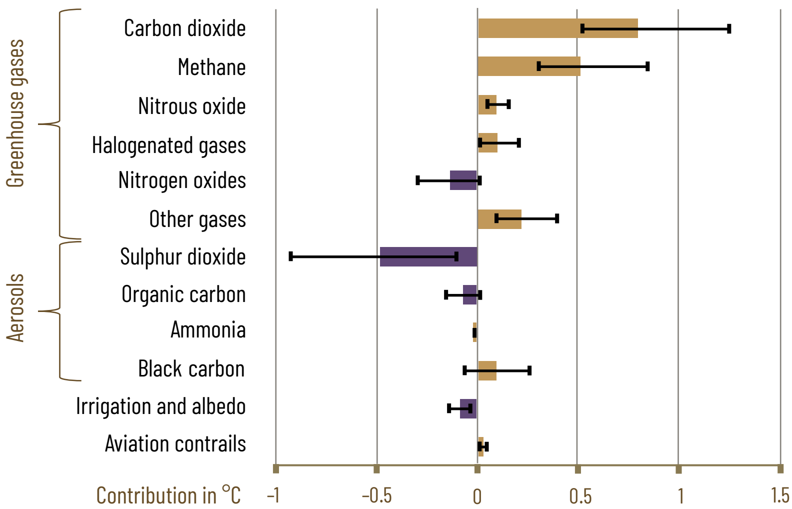
***Nitrogen oxides, $NH_3$ and aerosols:*** $NO_x$ and $NH_3$ can generate both warming and cooling effects on the climate through their influence on the atmospheric concentrations of $CH_4$, $O_3$ and aerosols. For instance, $NO_x$ induces the formation of $O_3$, which is a powerful GHG with warming effects, whereas $NO_x$ removal of $CH_4$ through the production of hydroxyl radicals ($OH^-$) causes cooling effects through both $CH_4$ and $O_3$ reduction. Moreover, $O_3$ can remove $CH_4$ through increasing $OH^-$, which generates a small cooling effect on climate. At the same time, both $NO_x$ and $NH_3$ enhance the formation of light-scattering aerosols in the atmosphere that reflect light back into space and cause a cooling effect (Pinder *et al.*, 2012). This effect is short-lived because aerosols have a relatively short atmospheric lifespan compared to $N_2O$ and other GHGs. Readers are referred to Lasek and Lajnert (2022) and Pinder *et al.* (2013) for an expanded discussion of climate warming and the cooling of $NO_x$.

***Nitrogen stimulation of non-$N_2O$ GHG:*** Nitrogen input to aquatic and terrestrial ecosystems can stimulate the emissions of non-$N_2O$ GHG such as $CO_2$ and $CH_4$, which contribute to radiative forcing, which denotes changes in the energy balance of the Earth's system due to increases in GHG concentration and other perturbations (Ramirez-Corredores *et al.*, 2023). Since 1750, these gases have largely dominated the overall warming influence on the Earth's climate (US EPA, 2016) (Figure 25). Methane is more potent than $CO_2$ but has a shorter atmospheric lifespan and is estimated to have a GWP of 27–30 times that of $CO_2$ over 100 years. While $CH_4$ lasts about a decade on average in the atmosphere, $CO_2$ lasts from 300 to 1000 years and accounts for a 36 percent increase in radiative forcing since 1990.

In aquatic ecosystems, N regulates GHG dynamics by influencing primary production and respiration (Cole *et al.*, 2000), methanogenesis (Bogard *et al.*, 2014), and $CH_4$ oxidation (Deutzmann *et al.*, 2014). Taking methanogenesis as an example, excess N often leads to hypoxia and anoxia in ocean and surface waters, which promotes the release of $CH_4$. Furthermore, episodes of hypoxia and anoxia can stimulate $CO_2$ emissions (Li *et al.*, 2017). These complex interactions often generate a warming effect on the global climate.

In most terrestrial and wetland ecosystems, N can affect $CH_4$ flux to the atmosphere, which is regulated by the balance between $CH_4$ production and consumption. Liu and Greaver (2009) performed a metanalysis of 313 observations and found that N inputs by deposition and fertilization increased $CH_4$ emission by 97 percent, reduced $CH_4$ uptake by 38 percent, and increased $N_2O$ emission by 216 percent when averaged across grasslands, wetlands and anaerobic agricultural systems. The influence of N inputs on $CH_4$ flux was only significant in anaerobic agricultural ecosystems. In the same study, N inputs increased C sequestration in forests and other ecosystems, but this $CO_2$ reduction was estimated to be largely offset (53–76 percent) by N stimulation of global $CH_4$ and $N_2O$ emissions from multiple other ecosystems (Liu and Greaver, 2009). In California forests, the deposition of N has been observed to enhance C storage in vegetation and SOM. When N excess leads to ecosystem N saturation, the resulting soil acidification, aluminium (Al) toxicity, and base-cation leaching may outweigh the advantages of N fertilization, ultimately leading to forest decline and loss of stored C (Aber *et al.*, 1998; Bowman *et al.*, 2008).

FIGURE 25
**Physical drivers of climate change**




*Source:* Authors' elaboration based on **IPCC.** 2021. Summary for Policymakers. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou, eds. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge, UK and New York, USA, Cambridge University Press. http://doi.org/10.1017/9781009157896.001

## 4.4 EFFECTS OF NITROGEN ON TERRESTRIAL ECOSYSTEMS

On a global level, there are regions with too much and too little N. The following sections explore the effects of N on terrestrial ecosystems, outlining the dual challenges posed by excessive and deficient levels of N. First, the discussion focuses on ecosystems dealing with N abundance, dissecting N effects on different aspects of ecosystem functions, including plant growth and soil C sequestration, soil acidification, soil and plant biodiversity, air quality and pollution. Subsequently, ecosystems experiencing N deficiency that offer insights into the nuanced outcomes linked to insufficient N levels are discussed.

### 4.4.1 Ecosystems with too much nitrogen

***Plant growth***

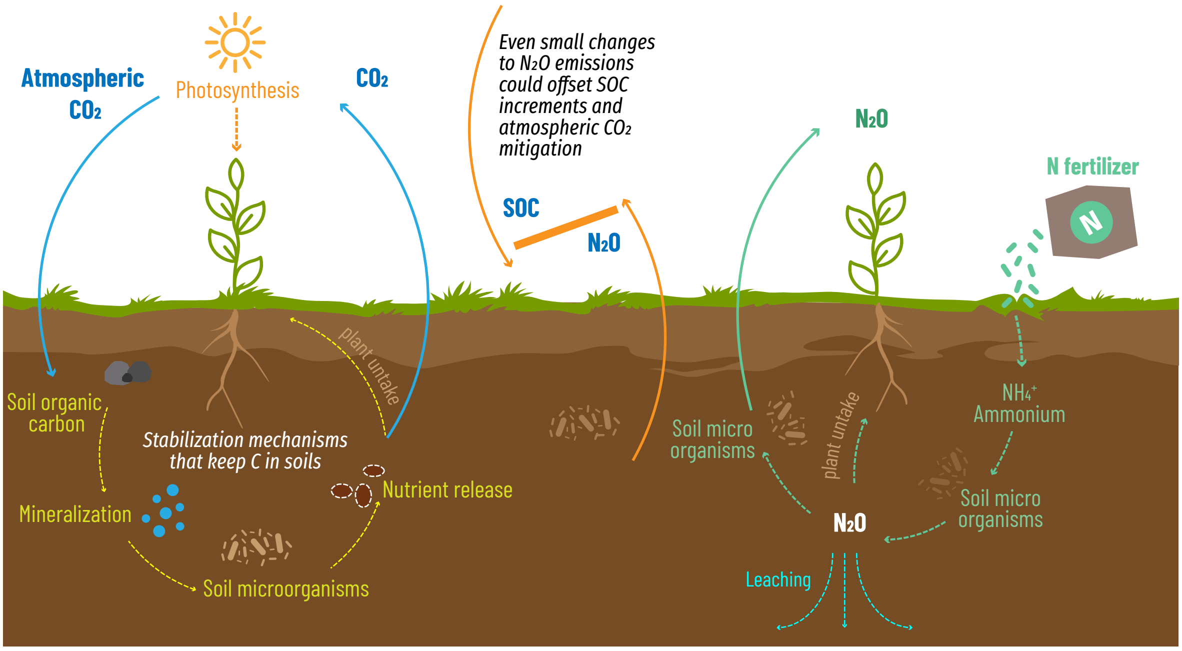
Because ecosystem productivity is often limited by N availability, N addition to soil, emanating from the use of N fertilizers and deposition, has markedly increased net primary productivity (Greaver *et al.*, 2016), improved agricultural production, and contributed to human and animal nutrition and well-being. High rates of N addition may decrease plant growth by increasing the concentration of soil acid anions (Greaver *et al.*, 2016), depletion of soil nutrients, reduction in SOM content, and degradation of soil structure. Excessive application of synthetic fertilizers has been shown to acidify soils over time and, as a result, reduce yields below their potential optimal levels.

***Effects of N on terrestrial C sequestration***

Because C and N cycles are tightly coupled in soils, agricultural management aiming at increasing soil organic carbon (SOC) will affect N cycling as well. Many studies showed that N additions are necessary to increase soil C stocks and sequestration (Greaver *et al.*, 2016; Li *et al.*, 2017; Lu *et al.*, 2021; Xia and Wan, 2008). For instance, Geisseler and Scow (2014) found that the addition of mineral fertilizers significantly increased organic C content by an average of 13 percent compared to unfertilized fields. Furthermore, Ganeshamurthy (2009) found that the continuous cultivation of pulses or food legumes increased SOC content because of their ability to fix $N_2$ and greater below-ground biomass. Nitrogen-induced SOC occurs through multiple mechanisms (Guenet *et al.*, 2021), all of which dictate that N is inevitably required to stabilize SOC and build up SOM.

These benefits may be offset by N-induced increases in $CO_2$ emissions (Grandy *et al.*, 2006; Liu and Greaver, 2010; Lugato, Leip and Jones, 2018) (Figure 26). Davies *et al.* (2021) demonstrated that with the achievement of C sequestration goals, N additions would generate an estimate of 1.7–2.5 Tg $N_2O$-N/yr in emissions. This would result in a reduction of the emission reduction potential of C sequestration by 213–319 Tg $CO_2$-C/yr. Thus, C sequestration cannot be achieved without considering the role of N (van Groenigen *et al.*, 2017). Carbon and N coupled interactions, including N originating from organic sources or through atmospheric N fixation, need to be integrated into SOM

FIGURE 26
**Potential trade-offs of N and C in crop systems**




*Source:* Authors' elaboration.

and climate change models to prevent over-estimation of C sequestration, particularly from $CO_2$ fertilization.

### ***Soil acidification***

Nitrogen-induced soil acidification (i.e. decrease in pH) stands as one of the top ten threats to soil health (FAO and ITPS, 2015). This pervasive issue affects approximately 30 percent of the world's ice-free land, with acidic soils covering a substantial portion of areas with agricultural potential (Sumner and Noble, 2003). Acidification is a pressing global challenge, with N addition estimated to cause a significant average reduction in soil pH of 0.26 on a global scale (Tian and Niu, 2015). The majority of the world's acid soils occur in Asia, Africa and the Americas. Low-income countries are the most affected because conventional means to buffer soil pH decrease are limited (FAO and ITPS, 2015).

Soil acidification occurs when acidity-generating processes offset acidity-consuming processes. It originates from increasing acidic precipitation and atmospheric N deposition of acidifying gases or particles and soil contamination. The most important cause of soil acidification in agricultural land is the misuse and overuse of fertilizers along with the application of $NH_4$-based fertilizers and urea, elemental sulphur fertilizer, and leguminous pastures (FAO and ITPS, 2015). Ammonium-based fertilizers account for 72 percent of the total N fertilizer input worldwide (IFA, 2020). Upon application, $NH_4^+$-based fertilizers and urea undergo nitrification and hydrolysis, releasing protons into the soil solution, which leads to a reduction in soil pH. The nitrification-driven acidification rate is approximately 10–100 times faster than that of acid deposition (Guo *et al.*, 2010). Compared to $NH_4^+$-based fertilizers, $NO_3^-$ sources are less acid-forming; for instance, ammonium nitrate ($NH_4NO_3$) generates less acidity per unit N compared to ammonium sulphate [$(NH_4)_2SO_4$] because only half of the N contained in the former fertilizer is oxidized (Strawn, Hinrich and O'Connor, 2020). Moreover, soil pH decreases linearly with increasing N rates (Dal Molin, Ernani and Gerber, 2020; Guo *et al.*, 2010; Tian and Niu, 2015). Another large effect on acid formation is the leaching of $NO_3^-$ below the root zone, which occurs with the removal of bases to maintain charge neutrality. Environments with high leaching potential are more acidifying (Weil and Brady, 2017).

High acid soil content can severely restrict plant growth and cause plant physiological changes and tree mortality. Acidity affects the availability of metals and nutrients, especially phosphorus, calcium, molybdenum and magnesium (Weil and Brady, 2017), and induces aluminium and magnesium toxicity (Zheng, 2010), thereby affecting plant health. Acidification contributes to the exacerbation of soil-borne diseases (Zhang *et al.*, 2022) by promoting pathogenic organisms in agricultural soils at pH < 5.5 (Li *et al.*, 2017) and impacting the movement of soil contaminants (Weil and Brady, 2017). The effects of climate change on N-driven soil acidification are likely to be dependent on initial soil and ecosystem properties (Rengel, 2011). Rising temperatures and increased precipitation can boost crop biomass production, leading to higher removal of basic cations and unbalanced C and N cycles (Tang and Rengel, 2003). The removal of basic cations can be further increased by climate change-driven increases in organic matter degradation and nitrification rates, which potentially increase $NO_3^-$ leaching and associated base cations. As such, alkaline components from the soil's exchange complex are transferred into surface waters and groundwater, leaving acidified soils. Nitrification can outpace weathering, the main mechanism for base cations in the soil (Greaver *et al.*, 2016). Excess exposure of soil to $CO_2$ can decrease aqueous pH by one to three units in soil pore water. Compared to well-buffered systems, poorly buffered systems, such as sandy soils, have less alkalinity-producing minerals and cannot resist changes in pH. In these systems, soil acidification and induced disturbances, exacerbated by climate change, are more pronounced. Particular attention should be paid to such systems (Hickman *et al.*, 2020; Ngatunga *et al.*, 2001). The effects of climate change on N-driven soil acidification are subject to debate, with some studies showing increases in soil weathering with rising precipitation and temperatures, which could eventually mitigate soil acidification. Eventually, the balance between weathering and nitrification in each system would dictate if acidification was aggravated or mitigated by climate change (Greaver *et al.*, 2016).

### ***Air quality and pollution***

Air pollution and climate change are deeply connected because the chemical species that contribute to the degradation of air quality are often GHG or co-emitted with GHG. As mentioned earlier, N emissions stimulate non-$N_2O$ GHG emissions and contribute to the formation of $O_3$ and PM2.5 (fine particulate with an aerodynamic diameter $\leq$ 2.5 micrometres), which together with $NO_2$ and $N_2O$, are major air pollutants with significant impacts on human health and crop productivity. These pollutants are a leading risk factor for premature mortalities worldwide, according to the Global Burden of Disease study (IHME, 2021), surpassed only by high blood pressure, tobacco use and poor diet.

Air pollution by N is accelerated by climate change, which promotes wildfires and emissions from terrestrial ecosystems. More wildfires are expected, which leads to increasing GHG emissions and PM concentrations. Changes in temperature and precipitation patterns are projected both to lengthen the $O_3$ season and intensify $O_3$ episodes in some areas. Warming temperatures increase the adverse health effects related to $O_3$ pollution (Jacob and Winner, 2009), such as respiratory infection and an increase in sensitization to allergens. Moreover, mitigating air quality degradation will become more challenging with higher air temperatures, as greater reductions in $NO_x$ emissions will be required to achieve equivalent reductions in $O_3$ pollution under warmer conditions (Wu *et al.*, 2020).

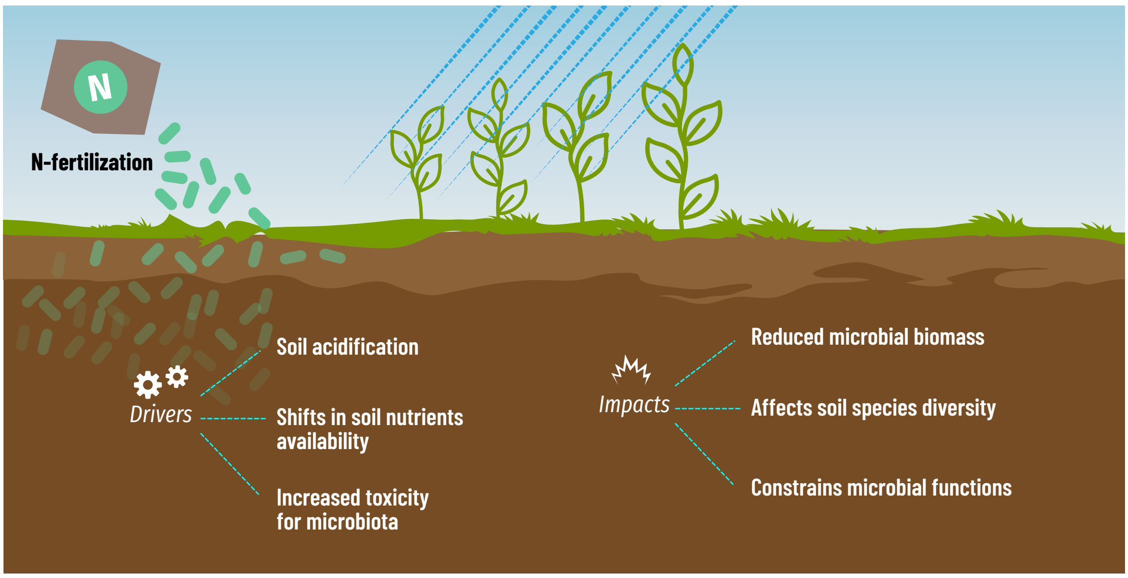
***Soil and plant biodiversity***

FAO *et al.* (2020) define soil biodiversity as "the variety of life belowground, from genes and species to the communities they form, as well as the ecological complexes to which they contribute and to which they belong, from soil micro-habitats to landscapes". Decades of research show that excess N addition reduces soil and plant biodiversity. Direct toxicity, soil acidification, nutrient imbalances, and interspecific competition caused by long-term N application are linked to plant biodiversity loss in many natural ecosystems (Dise *et al.*, 2011; Maskell *et al.*, 2010) (Figure 27). Furthermore, a meta-analysis across field studies subjected to N additions by Treseder (2008) showed that N fertilization reduced soil microbial biomass by 15 percent on average regardless of fertilizer types, ambient N deposition rates, or methods of measuring biomass. Several mechanisms were proposed to explain the decline of microbial biomass, including soil acidification. Nitrogen-induced soil acidity affects species diversity through habitat loss, constraints on important microbial functions, shifts in nutrient availability, and increased toxicity for microbiota, consequently altering soil food webs (FAO *et al.*, 2020). Moreover, the sensitivity of biota can be altered in the presence of climate change stressors, but it is currently unclear how climate change affects the biological thresholds of acidity (Greaver *et al.*, 2016). Other mechanisms include elevated osmotic potential and potentially toxic concentrations of the N forms added. Nitrogen-induced degradation of air quality and pollution can decrease plant biodiversity by increasing their exposure to pollutants. For example, $NH_3$ and $NO_x$ air pollution have led to reduced forest biodiversity by more than 10 percent over two-thirds of Europe due to increases in the growth of algal slime that can suffocate tree-living plants such as mosses and lichens, among other threats (Sutton *et al.*, 2011). There is evidence that suggests that plant biodiversity can recover following abatement of N deposition (Storkey *et al.*, 2015).

### 4.4.2 Ecosystems with too little nitrogen

While the reduction of anthropogenic N inputs to the Earth system is widely recognized as a high priority, many areas of the world, including much of sub-Saharan Africa, parts of Asia, and Latin America, frequently receive less than 50 kg N/ha compared with applications of 200 kg N/ha in developed countries (Dobermann *et al.*, 2022). This situation is troubling for the health, productivity and function of ecosystems and human beings. An example is seen in Africa, where small-scale farmers have limited access to N inputs and, as a consequence, have severely depleted soil fertility over the past decades (Wise, 2021). This depletion occurred as they cultivated crops relying on the soil N pools without adequately replenishing them using organic or synthetic fertilizers or other more sustainable practices such as crop rotations with leguminous plants, agroforestry, and the use of organic amendments, for example, compost

FIGURE 27
**Nitrogen fertilization drivers and impacts on biodiversity**




*Source:* Authors' elaboration.

and green manures (Valenzuela, 2023). Consequently, food accessibility is restricted for about 180 million Africans (Sanchez, 2002). The average annual depletion rate of N was estimated to have reached 22 kg/ha of cultivated land over the previous 30 years in 37 African countries (Sanchez, 2002). This not only hampers their ability to live healthy and productive lives but increases their vulnerability to socio-ecological shocks. Because low soil N availability restricts plants' growth and diminishes the N content of their leaves, cattle feeding on them have less protein in their diet, which decreases animal health and productivity (Craine, Elmore and Angerer, 2017). Elevated atmospheric $CO_2$ worsens the nutritional quality of food. Plants can exhibit reduced protein content due to increases in C:N ratios (Ebi and Loladze, 2019). Replenishing soil fertility for current and future generations cannot be solely achieved by relying on manure and legumes, despite their ability to provide a portion of N for crop needs (Houlton *et al.*, 2019). Synthetic fertilizers can address N deficiencies in these areas, but their high cost limits their accessibility by farmers. This calls for deep socioeconomic and political transformations to scale up sustainable fertility replenishment practices, build soil health and ensure the accessibility of N to underdeveloped nations.

## 4.5 EFFECTS OF NITROGEN ON AQUATIC ECOSYSTEMS

In the following sections, the effects of N on aquatic ecosystems through two important mechanisms are described: N-driven eutrophication and acidification. The interaction of these mechanisms with climate change and pollution of $NO_3^-$ in groundwater are described.

### 4.5.1 Eutrophication

Eutrophication is pervasive in many bodies of water, including freshwater, estuaries and coastal ecosystems (Le Moal *et al.*, 2019) (Figure 28). It is one of the most visible and ecologically significant consequences of increased N and P loadings. Because many primary producers are N-limited, N enrichment of aquatic ecosystems often alleviates this limitation, thereby increasing primary production and leading to the development of opportunistic algae blooms. Large blooms outcompete other species, leading to changes in algae community structure and species abundance. Consequently, fast-growing species with high N assimilation efficiencies are promoted (Nixon, 1995; Smith, 2003). Some of these species, such as dinoflagellates and diatoms, release toxins in the waters, causing mass mortalities of fish, mammals and birds, and affecting the quality of drinking water, compromising food security in certain regions and Small Island Developing States (Camargo and Alonso, 2006; FAO, IOC and AIEA, 2023; Gilbert *et al.*, 2006; Griffith and Gobler, 2020). This phenomenon is known as harmful algal blooms. When blooms die off, algal-derived matter that settles in bottom waters is subjected to the combined processes of sedimentation and microbial decomposition, which lead to the depletion of oxygen. These processes establish hypoxic and anoxic areas in open ocean and coastal waters, commonly referred to as "dead zones". These zones have been doubling in number and size since the mid-1990s (Breitburg *et al.*, 2018; Maúre *et al.*, 2021) and are exacerbated by warming temperatures, other climatic drivers (Altieri and Gedan, 2015) and anthropogenic activities (Nwankwegu *et al.*, 2019).

Increased competition from algal blooms diminishes native biodiversity in aquatic ecosystems at all trophic levels (Grizzetti *et al.*, 2011). For instance, Waycott *et al*. (2009) estimated that 29 percent of seagrass cover has declined in the past century. Changes from seagrass to ephemeral macroalgae caused by nutrient enrichment in shallow coastal areas may cause a loss of habitat for aquatic animals (Burkholder, Tomasko and Touchette, 2007). This ecological shift in native ecosystem biodiversity is further intensified by the creation of dead zones, which are inhospitable for most aquatic life, including fish, invertebrates and perennial underwater vegetation (Smith, 2003). Moreover, the lack of oxygen in these zones leads to the proliferation of hypoxia- or anoxia-tolerant species (Do Rosário Gomes *et al.*, 2014). The loss of native biodiversity leads to simplified and less-resilient aquatic ecosystems compared to their once-diverse counterparts. Due to the loss of biodiversity, various ecosystem services are affected by eutrophication (Kermagoret *et al.*, 2019), such as providing quality water for consumption and important economic activities, including fisheries, recreation, tourism, and aesthetics (Griffith and Gobler, 2020). Eutrophication causes water to have an unpleasant taste and odour, while becoming highly turbid and rich in nutrients, which in turn affects these eco-services.

The loss of ecosystem services is expected to intensify with climate change exacerbating water eutrophication (Nazari-Sharabian, Ahmad and Karakouzian, 2018). Alterations in precipitation patterns and increases in temperature increase nutrient loadings in aquatic habitats and create conditions that promote the proliferation of cyanobacterial blooms. The latter are expected to have a competitive advantage under future warming compared to other phytoplankton species, indicating a likely increase in bloom growth rate and their ability to attain larger sizes (Elliott, Jones and Thackeray, 2006; Jöhnk *et al.*, 2008).

Eutrophication affects climate change through GHG emissions. Projected N loadings and an increase in nutrient imbalances (such as N:P ratio) suggest a potential increase in future $N_2O$ emissions from aquatic ecosystems (Galloway *et al.*, 2008; Kumar, Yang and Sharma, 2019; Wang *et al.*, 2020). Eutrophic and hypoxia-affected areas promote the production of $N_2O$ from both nitrification and denitrification processes (Codispoti, 2010). Moreover, the high flux

FIGURE 28
**Eutrophicated Tota Lake in Colombia**


of $N_2O$ is produced by algae in eutrophic waters as part of their metabolism (Burlacot *et al.*, 2020). Hypoxic conditions, combined with the availability of quality organic C substrates, promote $CH_4$ production and emissions (West, Creamer and Jones, 2016). Algal blooms enhance $CH_4$ by rapidly exhausting the dissolved oxygen content of water, reducing the rate of $CH_4$ oxidation, and increasing the diffusive flux of $CH_4$ (Yan *et al.*, 2017). In wetlands, the promotion of vascular plant growth by excess N potentially raises the availability of C substrates for methanogenesis and facilitates the diffusion of $CH_4$ through plant stems (Erisman *et al.*, 2008).

Eutrophication has different impacts on $CO_2$ emissions. For example, the shift towards the dominance of algae and loss of macrophytes can increase $CO_2$ consumption and enhance rates of C sequestration (Kastowski, Hinderer and Vecsei, 2011). When algal blooms die and decompose, organic C mineralization of algal-derived matter releases $CO_2$ into the waters and potentially the atmosphere. The release of $CO_2$ can cause acidification by reducing water pH, which has implications for the carbonate system dynamics, such as decreases in dissolved $CO_2$ concentrations. An important subject of debate is whether the reduction in $CO_2$ emissions will be offset by increased emissions of $CH_4$ and $N_2O$ in aquatic ecosystems (Grasset *et al.*, 2020; Vachon *et al.*, 2020). Eutrophication and climate change can strengthen each other, creating positive feedback loops (Li *et al.*, 2017), which affect the functioning and resilience of natural ecosystems and increase mitigation costs (Halpern *et al.*, 2007; Su *et al.*, 2019). Global aquaculture of aquatic animals in inland waters produced 59.1 million tonnes in 2022, accounting for 62.6 percent of the total world aquaculture production. Moreover, 11.3 million tonnes were caught in 2022 – in addition to 1.3 million tonnes of algae (FAO, 2024a). Eutrophication can put this production at risk threatening food security and nutrition.

### 4.5.2 Acidification

Aquatic ecosystem acidification is another consequence of N loss via atmospheric N deposition. When emitted, $NO_x$ undergoes several chemical transformations in the atmosphere (Jacob and Winner, 2009), after which they are deposited into catchments and onto water surfaces as dissociation products of nitric acid ($HNO_3$). The latter is a strong acid that fully dissociates in water, releasing hydrogen ions ($H^+$). Ammonium deposition can contribute to acidity, as both the biological uptake of $NH_4^+$ and nitrification produce $H^+$. Nitrogen deposition can induce N saturation of terrestrial ecosystems and subsequent leaching of $NO_3^-$, carrying with it a loss of base cations, mainly calcium. Nitrogen deposition can mobilize inorganic Al. These leaching processes usually result in lower pH in both soil and waters, higher concentrations of Al associated with less P availability, and a reduction in acid neutralizing capacity of water bodies (Kopáček *et al.*, 2001; Shao *et al.*, 2021). Base cation losses from soil can buffer waters against the impacts of deposition; ultimately, base cation inputs into the lakes will decrease as exchangeable base cation pools become depleted (O'Dea *et al.*, 2017).

Nitrogen-induced acidification has initiated a cascade of negative environmental effects on aquatic ecosystems, with detrimental impacts on societal uses, fisheries resources, and economies. The most direct effect is on aquatic biodiversity. Persistent acidification shifts species composition at the base of the food chain and favours acid-tolerant macrophytes and phytoplankton. Moreover, acidification reduces species diversity (Sunday *et al.*, 2017) through multiple mechanisms, such as decreasing carbonate ion concentrations which impact marine calcifying life (Doney *et al.*, 2009) and increased toxicity of aluminium ion ($Al^{3+}$) and hydrogen to fish and aquatic invertebrates (Baker *et al.*, 1990; Cosby *et al.*, 2006). Lakes with lower pH and acid-neutralizing capacity in the Adirondacks cannot support fish (Gallager and Baker, 1990). These shifts in fish abundance and diversity have implications for sport fishing, recreation, and cultural and existence values (Banzhaf *et al.*, 2006). Effects of acidification have been reported on riverine birds and amphibians (Durand *et al.*, 2011; Ormerod and Durance, 2009).

It is well established that acidification regulates water-to-atmosphere fluxes of trace gases. For instance, enhanced $N_2O$ consumption and reduction in nitrification and $NH_3$ oxidation rates in acidified seawaters have been reported in many studies (Beman *et al.*, 2011; Zhou *et al.*, 2023). Conversely, higher nitrification rates (which could increase $N_2O$ production) in some coastal and estuarine waters were reported and attributed to low pH-adapted nitrifier communities (Fulweiler *et al.*, 2011). Some recent studies noted an increase in $N_2O$ production as a by-product during nitrification through "hybrid" mechanisms associated with aquatic ammonia-oxidizing microorganisms and by alteration in microbial diversity (Frame *et al.*, 2017; Wu *et al.*, 2020; Zhou *et al.*, 2023). Ocean waters are not a significant source of atmospheric $CH_4$ (Wuebbles, 2002), but there is potential for direct impacts of acidification on methanogenesis via two production pathways (Repeta *et al.*, 2016). Some recent studies show that pH may alter the microbial communities responsible for $CH_4$ cycling in coastal sediments (Reshmi *et al.*, 2015) and by enhancing $CH_4$ oxidation due to heavy metal mobilization (Brocławik *et al.*, 2020). More research on the effects of acidification on GHG consumption and production from aquatic ecosystems is encouraged, as this research will create better-informed management programmes and more accurate GHG budgets for these systems.

While acidification regulates climate change through effects on GHG emissions, acidification is affected by climate change. Variation in rainfall and temperature patterns, attributed to climate change, disturbs the wet and dry cycle that regulates acidic inputs in precipitation, soil microbial processes, and the release of acid anions into surface waters. During drought, the soil accumulates acid anions, and when rainfall eventually occurs, the eventual flushing can trigger intense episodes of acidification (Greaver *et al.*, 2016). This abrupt rise in acidity poses significant threats to ecosystems. If these acidification events coincide with the presence of vulnerable organisms or crucial life stages, the repercussions become even more severe (Greaver *et al.*, 2016). Understanding how acidification and climate change can strengthen each other is crucial to implementing appropriate response strategies.

### 4.5.3 Inland waters: nitrate pollution in groundwater

Worldwide, groundwater accounts for over a third of the total water supply (Famiglietti, 2014), providing almost 50 percent of drinking water and 40 percent of irrigation water (Abascal *et al.*, 2022).

Nitrogen contamination of this essential resource is a significant concern that will be amplified by climate change. Nitrogen contamination mainly comes from the overuse of N fertilizers in agriculture, over-irrigation, animal husbandry, wastewater discharge, and landfill leachates. Specifically, $NO_3^-$ is highly soluble and can easily infiltrate the soil and leach into the groundwater. Elevated $NO_3^-$ levels in drinking water can pose health risks, particularly for vulnerable populations such as infants and pregnant women. Nitrate is believed to compromise the oxygen-carrying capacity of blood, a condition known as methemoglobinemia (or blue baby syndrome). Apart from compromising drinking water quality, $NO_3^-$ in groundwater can be discharged into surface water streams, causing eutrophication. In response to this challenge, FAO recommends a maximum of 22 mg per litre of $NO_3^-$ in drinking and irrigation water (Misstear, Banks and Clark, 2017). When groundwater is used for irrigation, $NO_3^-$ concentrations above this threshold were found to affect crop yields depending on crop sensitivity to $NO_3^-$. This threshold is tighter than the World Health Organization threshold currently at 55 mg/l.

The vulnerability of groundwater to $NO_3^-$ contamination is a function of agricultural management (e.g. quantity, rate, timing, methods of N application, irrigation), hydrogeologic factors (e.g. hydrologic properties of soil, permeability and thickness of the vadose zone, amount, timing and location of aquifer recharge), precipitation, temperature and characteristics of the terrain (Abu-alnaeem *et al.*, 2018; Roy *et al.*, 2020; Varol and Şekerci, 2018), land use and groundwater table fluctuation (Machiwal *et al.*, 2018; Zhang and Furman, 2023). Addressing these factors is essential for effective groundwater management and the design, enforcement and monitoring of regulatory policies (Wick, Heumesser and Schmid, 2012). With that regard, developing groundwater vulnerability maps, groundwater vulnerability index, and N-risk maps under varying hydrogeologic and hydro-climatic conditions can assist governments in establishing efficient policies (Machiwal *et al.*, 2018).

## 4.6 APPROACHES FOR ASSESSING THE IMPACTS OF THE NITROGEN CHALLENGE

Over the years, researchers and scientists have shown a growing interest in understanding the role of N cycling in agricultural systems. As a result, approaches, methods, metrics and performance indicators have been developed to monitor, measure and evaluate N fluxes and impacts on ecosystems. The common framework for addressing N losses and pollution is characterizing the multiple N impacts, which are driven by N flows, using specific indicators and metrics. Models that integrate this information can be used. Based on this integration, practical solutions and appropriate N measures are developed. This section explores the state of knowledge alongside the challenges, limitations and opportunities on these subjects, although this exploration is not exhaustive.

### 4.6.1 General nitrogen assessments

Assessments are an increasingly common method researchers use to analyse existing data sets across multiple scientific disciplines and obtain a comprehensive overview of established knowledge and areas of scientific uncertainty. A notable example of such assessments is the global effort that resulted in reports by the Intergovernmental Panel on Climate Change (IPCC) (UNEP-WCMC, 2010). More recently, several regional N assessments have been conducted in North America, South Asia, and Western Europe, with an ongoing international N assessment as part of the Global Environment Facility/UNEP "Targeted Research for improving understanding of the global N cycle towards the establishment of an International Nitrogen Management System". These assessments provide detailed accounts of regional N drivers and flows, impacts on ecosystem health and human well-being, best practices and policy options, and their potential effects on agriculture, the environment, and human health if implemented. The consolidation of N data in an assessment aims to overcome data limitations and fragmentation across research, policy, and multiple data sources and scientific disciplines, which often restricts discussions, cooperative actions, and solution development. The US Environmental Protection Agency (US EPA) outlined a structured approach for developing a comprehensive, integrated science assessment (US EPA, 2015). This approach involves conducting thorough literature searches, rigorously evaluating the quality of individual studies, synthesizing and integrating evidence, and formulating scientific conclusions and causal determinations based on the findings. This approach adopts a participatory framework involving actors across the agrifood chain and relevant economic sectors to frame scientific questions, aggregate available information, and identify sources of uncertainty. Obtaining accurate and scientifically sound data remains a significant challenge. For instance, N fertilizer use is typically not comprehensively monitored at relevant scales, and data sources may be inconsistent. A notable example is the significant variation in fertilizer N use between researchers' and farmers' managed fields. In practice, fields managed by researchers often apply fertilizers at recommended rates for ideal management practices, whereas farmers may adapt their fertilizer usage based on cultural, economic and environmental considerations. Translating findings from laboratory experiments to larger scales, such as ecosystems, is challenging, potentially limiting the range of responses that can be adopted. Other challenges include the fact that the impacts of N on ecosystems often arise from a balance of synergistic and antagonistic influences, and actions taken to address one pollutant or N compound may inadvertently worsen another in "pollution swapping". Nitrogen assessments are general and do not adequately address N challenges under different soil, crop, water management and climatic conditions and scenarios. Moreover, such assessments have focused on managing surplus N and have not been conducted in areas of the world where N-depleted soils are a concern. Despite these challenges, N assessments are crucial for informing decision-making and promoting sustainable N management practices. Efforts should continue to enhance the accuracy and scope of these assessments to address evolving environmental and agricultural needs effectively.

### 4.6.2 Functional matrix to define and identify nitrogen effects on ecosystems

A functional matrix is a theoretical framework that could be used to identify the driving forces responsible for exerting pressures and impacting ecosystem functioning. It serves to identify both the direct and indirect pathways to N impacts, focusing on indicators that describe relationships between pressures to states and between states and impacts. An example of a functional matrix with drivers, pressures, state and associated impacts of N on ecosystems is provided in Table 4. There are some important considerations in building such a matrix; for example, the fact that a single pressure can lead to multiple impacts and that the matrix only considers pressures that make a substantial contribution to the observed impacts.

TABLE 4
**Examples of drivers, pressures and impacts in a functional matrix to assess N impacts on ecosystems**

| Drivers: human population and economic growth increase N demand for food, energy, goods and services | Pressures | Impacts |
|---|---|---|
| **Examples:** | Examples: | Examples: |
| **Organic and synthetic N fertilizer use on croplands** | GHG emissions to air | Climate and air quality |
| **Livestock, feed and manure management** | N input to surface and groundwaters | Surface and groundwater quality |
| **Land use, land cover and land management (including pasture and rangeland management)** | N input by deposition | Ecosystem biodiversity and C storage |

*Source:* Authors' elaboration.

### 4.6.3 Nitrogen budget, footprint and critical load approaches for impacts assessment on ecosystems

***Input–output N budgets***

Over time, N budgets have been developed at various scales, including Earth-scale (Johnson and Goldblatt, 2015), regional (Lin *et al.*, 2020; van Egmond, Bresser and Bouwman, 2002), national (Derwent, Dollard and Metcalfe, 1988), and within specific systems such as livestock systems (Oenema, 2006) and farm scale (Cherry *et al.*, 2012). Nitrogen budgets serve to identify N sources, sinks and flux magnitudes and determine if there is an N surplus or deficit, as well as the fate of N in the system studied over a specific time-period (Cherry *et al.*, 2012). Understanding N balance is crucial for evaluating N cycling performance and developing strategies for reducing N losses to the environment (McLellan *et al.*, 2018). Gross N balance is a commonly used indicator to assess N pressure from agricultural sources (European Commission, 2018) to track the size of N flows and determine the state of equilibrium or imbalance between N inputs and N outputs within a system.

***Nitrogen critical load***

The N critical load approach has primarily been used to describe the vulnerability of natural ecosystems to the atmospheric deposition of N (de Vries *et al.*, 2007). Nitrogen critical load is defined as "the amount of N deposition below which no significant effects to the ecosystem are thought to occur according to current knowledge and is meant to inform the amount of N an ecosystem may endure before unwanted effects become manifest" (Nilsson, 1988). It is usually dependent on the system studied and estimated using a combination of field studies and dose-response relationship parametrization. Nitrogen critical load has been a key science-based tool for assessing the environmental consequences of air pollution. This approach has been used as a policy tool and informed policy negotiations by both the European Union Commission and the Convention on Long-range Transboundary Air Pollution (CLRTAP), in which European maps of N critical load and critical levels have been used to reduce N emissions to air (Amann *et al.*, 2011).

***Nitrogen footprint and life cycle assessment***

Nitrogen footprint and life cycle assessment (LCA) methods can be utilized to evaluate the ecological impacts of N on ecosystems indirectly. Both methods provide a way to quantify N flows, which can be applied to quantify inputs into ecosystems and further assess the ecological impacts of N. The N footprint measures the sum of N losses resulting from human activities, acting as a proxy for the wide-ranging environmental and health effects of N pollution. Life cycle assessment encompasses two components, namely life cycle inventory (LCI), which tracks the flow of substances and energy, and life cycle impact assessment (LCIA), which quantifies the subsequent impacts on the environment and human well-being. Life cycle assessment employs a comprehensive array of impact indicators that do not overlap, enabling stakeholders to evaluate effectively trade-offs between various types of impacts. Perming (2012) used both methods to evaluate N species for global warming, eutrophication, acidification, photochemical $O_3$ formation, and stratospheric $O_3$ depletion in Swedish-grown tomatoes and found that N footprint is more similar to LCI in that it did not provide insights into the environmental spectrum of impacts caused by N species, except for eutrophication. This debate is still current among researchers. Some argue for modifying the N footprint to resemble more of an impact indicator because it is often interpreted as such. On the other hand, others caution against this approach because accurately accounting for N impacts is challenging and could introduce more uncertainty into the N footprint concept. For a more detailed discussion, the reader is referred to the analysis (Einarsson and Cederberg, 2019).

### 4.6.4 Metrics for nitrogen evaluation and monitoring

Nitrogen use efficiency is a critical metric for assessing N dynamics, as described in Chapters 2 and 3. Recently, the NUE concept has been expanded to include the entire food chain, extending beyond agriculture (Kanter *et al.*, 2020b).

The valuation of ecosystem services as affected by the N challenges has been the subject of many studies. Sobota *et al*. (2015) explored the development of ecosystem services accounting systems that quantify the costs of N damages to clean air, safe drinking water, fisheries, and mitigation measures. Implementing such systems globally can raise public awareness and support funding for agricultural conservation programmes that reduce N releases. Knowledge gaps, including the valuation of cultural services, hinder the comprehensive assessment of N impacts (Jones *et al.*, 2014). Continued research on N metrics capable of documenting the conditions of the N cascade and its impacts on ecosystem services is central to the development of response strategies. While metrics that address a single N source or impact are important, collective metrics, such as those used to define acidification (as $H^+$ equivalent) or report global warming and GHG emissions (as $CO_2$ equivalents), permit the comparison of different management practices that affect the environment through multiple pathways. Collective metrics are not currently available for each environmental impact. Moreover, impact trade-offs (where an N strategy or source improves one aspect at the expense of another) are not well reflected in the current metrics. Developing such metrics is essential, and their promotion should be accompanied by efficient strategies to enable policymakers to take them up and ensure they move beyond research papers into practical implementation.

## 4.7 CONCLUSION AND KEY MESSAGES

Increased N input in terrestrial ecosystems leads to N saturation and decreases the capacity of ecosystems to store N. It imbalances N and C inputs, which can lead to an increased conversion to $NO_3^-$ subjected to leaching and denitrification processes. Ultimately, this results in N entering aquatic ecosystems. Additionally, too much N from misuse and overuse of fertilizers leads to soil acidification, which can restrict plant growth and cause plant physiological changes, tree mortality, and plant biodiversity loss. Furthermore, N directly and indirectly affects the climate through emissions of $N_2O$, $NO_x$, $NH_3$ and aerosols, contributing to climate change and air pollution, which in turn affect ecosystems, biodiversity and human health.

Through leaching of N compound to water bodies, too much N causes eutrophication, which can lead to the development of algae and phytoplankton blooms, severely affecting native biodiversity. Acidification of water bodies affects aquatic biodiversity as well, posing risks for species diversity and a decrease in fish populations.

Conversely, ecosystems with too little N can be found in LMICs where agriculture has depleted soil fertility through inadequate fertility management due to low access to synthetic fertilizers and sound manure management. Soil N depletion severely affects soil health, leading to decreased crop and livestock production and food security while affecting the health and functioning of ecosystems.

Worldwide, N sits at the nexus of multiple social and environmental debates. Decision-makers, the scientific community, and relevant stakeholders are poised to act, particularly given the rise of concepts like natural-based solutions and circular N use, which have amplified interest in managing agricultural lands to enhance NUE and minimize N impacts on ecosystems. Finding and selecting appropriate solutions tailored to the specificities of the situation, which involves dealing with potential trade-offs with other N externalities, appears to be constrained by the lack of accurate information and uncertainties associated with the nature of the N cycle.

These challenges should not dissuade actions. Protecting the ability of ecosystems to provide services is the foundation of our lives and social systems. Countries and other agrifood system stakeholders are required to:

1. Advance the science of N by enhancing measurements of impacts, and N flows along the N cascade, and enabling research in techniques and approaches to improve N uptake and retention in crops and livestock.
2. Address farmers' perceptions, structural issues, and barriers to the adoption of N technologies and practices as part of projects and programmes of development.
3. Assist in context-specific recommendations and avoid one-size-fits-all prescriptions to increase NUE and deal with N losses to ecosystems.
4. Adopt an integrated approach to address N challenges.
5. Enhance collaboration among different stakeholders such as policymakers, governmental agencies, universities, growers, industry groups, public interest groups, environmental nonprofits and NGOs, consumers, and community members.
6. Minimize pollution swapping and trade-offs and enhance synergies by integrative planning of projects and programmes of development.

# Chapter 5
# Transforming agrifood systems through circular bioeconomy

## 5.1 INTRODUCTION

According to FAO, the bioeconomy is defined as the production, utilization, conservation and regeneration of biological resources, including related knowledge, science, technology and innovation, to provide sustainable solutions (information, products, processes and services) within and across all economic sectors and enable a transformation to a sustainable economy (FAO, 2021b). To do this, it capitalizes on unprecedented advances in life sciences and biotechnologies. Also, circular approaches can be used to improve the sustainability of the bioeconomy when it comes to resource use efficiency (Lang, 2022). The bioeconomy is not always sustainable, and it should assess context-specific principles and criteria (Gomez San Juan, Bogdanski and Dubois, 2019). In agrifood systems, circular bioeconomy approaches can support soil management and restoration, unavoidable waste valorization, and biomass use optimization. Livestock systems are one of these cases, where recycling, recovery, reuse, and other circularity principles are particularly relevant. These approaches do not solely fall under circular bioeconomy approaches, as their principles align with the principles of agroecology (FAO, 2018c). Hence, through agroecological approaches, sustainable development and N management in agrifood systems can be addressed as well.

The production of food needs a vast amount of resources (i.e. land, water, energy and labour). When food is lost or wasted, these resources, including N, are wasted, impacting the efficiency of food production. Recovering all the N, P and K from organic waste streams at the global level would be 2.7 times the nutrients contained within the volumes of chemical fertilizer currently used (Ellen MacArthur Foundation, 2019). The world produces enough food to feed the human population, yet millions suffer from hunger and malnutrition. Food loss and waste exacerbate this problem by reducing the amount of food available for consumption, contributing to food insecurity. Food items with high nutritional values, such as fresh produce or animal products (water- and land-based), are particularly impacted by high loss rates. Food loss and waste translate into a substantial economic loss. This impacts producers, consumers and nations as a whole and affects livelihoods and economic stability.

A recent FAO publication presents the bioeconomy as an opportunity to achieve SDG 12, Indicator 12.3, targets of reducing food losses (unspecified target) and reducing waste by 50 percent by 2030 (FAO, 2023d). The ten-year FAO Programme Priority Area "Bioeconomy for Sustainable Food and Agriculture" further addresses three targets of SDG 12:

- Target 12.2: By 2030, achieve sustainable management and efficient use of natural resources.
- Target 12.4: By 2020, achieve the environmentally sound management of chemicals and all wastes throughout their life cycle, in accordance with agreed international frameworks, and significantly reduce their release to air, water and soil in order to minimize their adverse impacts on human health and the environment.
- Target 12.5: By 2030, substantially reduce waste generation through prevention, reduction, recycling and reuse.

This chapter presents opportunities for the agrifood system within a circular bioeconomy to increase NUE and reduce N pollution from agricultural practices.

## 5.2 PRINCIPLES TO ENHANCE CIRCULAR BIOECONOMY IN AGRIFOOD SYSTEMS

### 5.2.1 Key elements

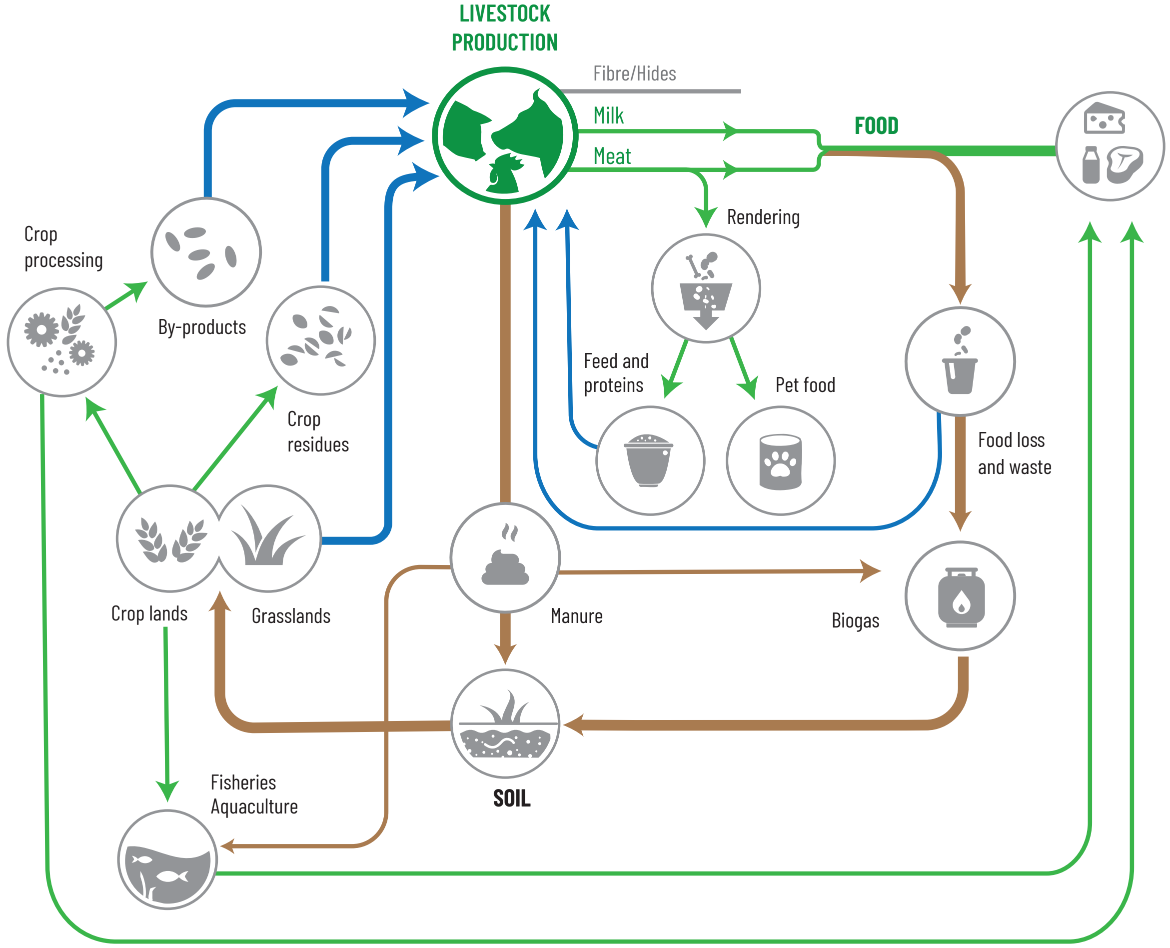
Circular bioeconomy approaches can enhance the sustainable development of the agrifood system. Its focus is on minimizing loss of nutrients to the environment and increasing the efficiency with which food is produced. De Boer and van Ittersum (2018) and van Zanten *et al.* (2019) propose the following principles of a circular agrifood system.

1. Food losses and waste should be avoided: through this, the availability of food can be increased, and losses of nutrients are minimized, enhancing food security and nutrition.
2. Inevitable FLW streams, as well as by-products from the agrifood chain, should be recycled back into the food system; through this, nutrients are used in the food system and loss of nutrients to the environment is minimized. Furthermore, it enhances the sustainable use of natural resources.
3. Arable land should be primarily used to produce food for direct human consumption; through this, the efficiency with which natural resources are used is maximized. This contributes to meeting the increased demand for food for a growing world population and, hence, increases food security and nutrition for all.

4. Use livestock to convert biomass unsuitable for human consumption to produce high-quality food: ruminants can be fed on grasses and other plants that humans cannot digest and convert these into milk and meat. Monogastric animals can be fed on waste streams and by-products that cannot be used for human consumption – producing meat, eggs and other products. Through this, livestock play a vital role in providing high-quality and nutritious TASF that contributes to a healthy diet and provides essential vitamins and minerals.

Livestock form an essential part of the agrifood system, and its development, based on the circularity principles above, varies between countries and regions. One-third of available agricultural land globally is suitable for crop production. Using this land to produce crops for direct human consumption maximizes the resource use efficiency of food production on these lands and increases the amount of food available for human consumption (Muscat *et al.*, 2021). As some livestock systems are heavily dependent on feed produced on these croplands, feed–food competition is prevalent in these livestock systems. By-products from the food processing industry, as well as FLW, can be converted by pigs and poultry into highly nutritious TASF. This would mean livestock systems dependent on concentrated feed produced on croplands have to transition to systems where by-products and organic waste streams are used as feed. In other regions, livestock unlocks biomass from marginal lands, which constitute two-thirds of global available agricultural land (Mottet *et al.*, 2017). This biomass that is unsuitable for human consumption can be used by livestock to convert human-inedible biomass to highly nutritious TASF, thus being used in a way that is compliant with the proposed circular principles (see Figure 29). It, therefore,

FIGURE 29
**Diagram that represents circular bioeconomy approaches in the livestock sector**




*Note:* The green arrows represent the production of food, blue arrows represent the supply of animal and fish feed, and the brown arrows represent the recycling flows across agrifood systems.

*Source:* Authors' elaboration.

depends on the livestock production system in what way it will need to transform to create a circular food system where nutrient loss is minimized and food is produced as efficiently as possible to feed the growing world population. A result can be that less livestock can be supported in certain regions, which would necessitate a shift to a more plant-based diet, increasing the amount of plant proteins. For example, a recent study by Simon *et al*. (2024) found that the integration of circularity principles in the European Union could reduce land use and GHG emissions by 44 and 70 percent, respectively. Current protein intake levels would remain the same, however livestock numbers would decrease overall.

By adopting circularity principles in agricultural production, N use within the production system is managed sustainably, as the priority is to recycle N within the system and minimize loss of N to the environment. Section 5.7.1 describes a case study focusing on minimizing N loss through on-farm circularity principles. By maximizing the efficiency with which food is produced, the overall NUE of the production system increases, too. Increasing circularity within the agrifood system, therefore, links to increasing the sustainable use of N.

Beyond crop and livestock production, other economic activities within agrifood systems are affected by and can contribute to the sustainable management of N. This section will not explore them in depth or provide examples, but it should be considered that all sectors can support effective N management, and it is only possible with a systems approach. For instance, fisheries and aquaculture can implement circular bioeconomy solutions for sustainable N management, which increases NUE and minimizes waste. In aquaculture, key opportunities to recycle N and minimize loss are through low-protein feeding, recovery of by-products, and recycling of nutrients from wastewater (Campanati *et al.*, 2022). For fisheries, increasing circularity can be achieved along the entire production and processing chain, focusing on minimizing losses and utilizing by-products and by-catches (Cooney *et al.*, 2023).

Not only are primary production or tertiary waste management activities important, but so are the food and bio-based processing industries. These are essential to implementing an effective bioeconomy that can alleviate environmental impacts. For instance, forest-based industries are applying circularity principles based on the cascading use of resources that maximize the use of wood and other forestry products (FAO and UNECE, 2023).

The bioeconomy is a strategy proposed by many countries to tackle climate and environmental issues (Gomez San Juan, 2024). For instance, in the European Union, the bio-based sectors (e.g. agriculture, forestry, bioenergy, biomaterials) have the potential to reduce GHG emissions by up to 2.5 Gt $CO_2$eq per annum by 2030 (European Commission, undated).

### 5.2.2 Options for circular management of nitrogen in the bioeconomy

FAO proposes implementing both upstream and downstream bioeconomy solutions to reduce N pollution. Figure 30 shows the different biomass value chain stages and how N can be incorporated. Circularity should be considered in upstream activities (e.g. agricultural production) and downstream (e.g. organic waste treatment facilities) and bioindustries. Taking an agrifood system approach can help better understand the natural resources and nutrients within a given system and optimize their production, processing and disposal.

Upstream solutions range from reducing the use of chemical and harmful products from fertilizers, pesticides, plastics, fuels, and other agricultural inputs (used in both food and feed production) by employing biological solutions and practices for the reduction of extensive and pollutant agriculture and the restoration of degraded and polluted ecosystems. This approach results in healthier soils and plants that require fewer inputs, increase NUE and improve human and environmental health overall. Bioeconomy innovations can also be implemented downstream, offering cost-effective and sustainable solutions to recover and recycle nutrients from effluent water in farms or agrifood industries to produce biogas, fertilizer, and a range of value-added products, as well as to restore polluted ecosystems with different soil and water remediation technologies.

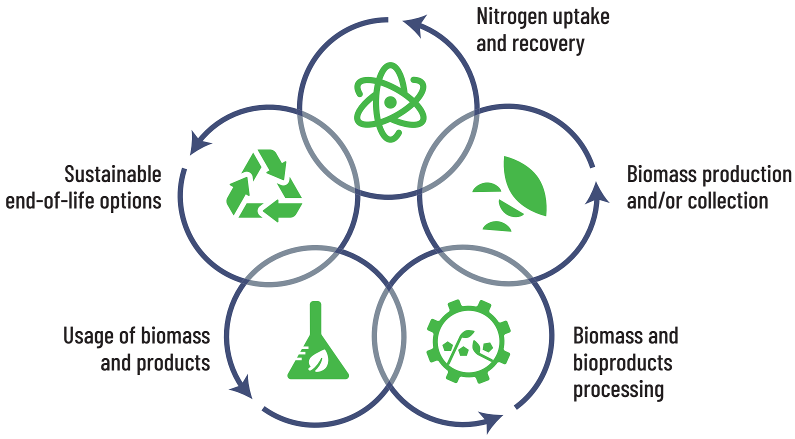
The following sections look at promising solutions in agricultural production, with a focus on circular practices at the production level.

## 5.3 PROMISING CIRCULAR BIOECONOMY SOLUTIONS IN AGRICULTURAL PRODUCTION

### 5.3.1 Efficient nitrogen management and ecosystem restoration through microbial and bio-based solutions

Excessive accumulation of N in the environment poses a threat to waterbodies from eutrophication and hypoxia, stimulating the growth of aquatic life typically resulting in dissolved oxygen depletion (see Chapter 4). Around 75 percent of the global ocean and freshwater eutrophication is caused by agriculture (Poore and Nemecek, 2018). A particularly common source is the high volume of animal manure that is not managed properly, together with over-fertilization. Emission prevention and control of N, improved wastewater management, and nutrient recycling from waste streams are measures to address various forms of N pollution. Sustainable bioeconomy practices aim to support N management in balancing efficient production and consumption while avoiding waste and inefficient use. Improving NUE in agriculture, increasing reliance on biological N sources as opposed to synthetic fertilizers, and implementing strategies to reduce loss and waste can enhance sustainable N management. FAO has identified

FIGURE 30
**Nitrogen use and stages of the biomass value chain**




*Source:* Authors' elaboration based on **Gomez San Juan, M., Bogdanski, A. & Dubois, O.** 2019. *Towards sustainable bioeconomy – Lessons learned from case studies.* Environment and Natural Resources Management Working Paper 73. Rome, FAO. https://openknowledge.fao.org/handle/20.500.14283/ca4352en

the most promising bioeconomy solutions to support the mitigation options of the IPCC and biofertilizers support to reduce $N_2O$ emissions (Gomez San Juan, Harnett and Albinelli, 2022a). The IPCC Mitigation and Adaptation via the Bioeconomy explains the relevance of the bioeconomy for climate change mitigation and adaptation with examples such as cellulose-based textiles to replace cotton or other textiles that require large amounts of chemical fertilizers and pesticides (IPCC, 2022).

The bioeconomy can furthermore ensure a healthier soil microbiome that supports N management (Kendzior, Raffa and Bogdanski, 2022). A review by FAO has established a direct correlation between crop production, soil microbiome, and climate change effects (Kendzior, Raffa and Bogdanski, 2022). Efficient microorganisms can enhance organic matter utilization, boost P solubility, and facilitate N fixation, mitigating GHG emissions by up to 10 kg $CO_2$eq/kg mineral N replaced. Additionally, adequate approaches for N pollutant remediation, such as microbial- or phytoremediation to remove excess nutrients from soil and water, could help recover N that can be used in – and substitute – a range of products, including food, feed, agricultural inputs and feedstock (Gomez San Juan, Harnett and Albinelli, 2022b). Different residues and waste can be used for bioremediation techniques to treat eutrophication, including techniques for biological N and P removal and recovery, such as sugarcane bagasse, coir pith, eggshell, wood, orange peel, and soybean milk residues (El-Sheekh *et al.*, 2021).

Some major benefits associated with biofertilizers are N fixing, promoting root growth, promoting yields, allowing nutrient uptake in acidic soils, and reducing the need for environmentally damaging pesticides and fungicides. For instance, using N-fixing inoculants can lead to average yield increases of 20–30 percent (Kendzior, Raffa and Bogdanski, 2022).

Microorganisms improve the efficiency with which organic matter is utilized by plants. Some of these (e.g. arbuscular mycorrhizae and rhizobacteria) increase N fixation in the soil and soil N uptake by the plant (Hack *et al.*, 2019). Azolla and phosphobacteria are used as microbial inoculants in soils that reduce methane emissions substantially and enhance NUE. Furthermore, some microorganisms (e.g. *Azospirillum, Bacillus, Pseudomonas, Trichoderma*) can play a vital role in promoting plant growth, biological control, enhancing N and water use efficiency, and mitigating the effect of biotic stress. Application of a microbial inoculant can reduce the reliance on synthetic fertilizers by 25 percent while maintaining crop yield (Gaspareto *et al.*, 2023; Mourouzidou *et al.*, 2023).

Policy is key to raising awareness of the benefits of biofertilizers and the use of microorganisms. Providing advice and consultation on bio-inputs can help enhance their implementation. This has been done, for example, by Argentina's Ministry of Agriculture, Livestock and Fisheries, which set up an Advisory Committee on Bio-Inputs for Agricultural Use. In combination with the creation of bio factories at the provincial level, the government supports the development and production of different biopreparations (Gomez San Juan, Harnett and Albinelli, 2022b).

### 5.3.2 Upcycling food, feed and waste loss

In addressing the challenges posed by traditional fertilizers, the agricultural sector must adopt innovative solutions that prioritize sustainability and environmental responsibility. The overuse of N, P and K formulations, as well as high livestock densities, have led to nutrient imbalances, environmental degradation, and soil health concerns. On the other hand, globally, annual production includes 125 million tonnes of N from livestock manure and 140 billion tonnes of lignocellulosic (i.e. biomass) wastes, constituting 30–40 percent of the overall solid waste output. This poses a global challenge for the management and utilization of livestock manure, agricultural waste and by-products. In certain countries, the recycling rate for total manure and agricultural crop residues ranges from 30 to 75 percent (Greff *et al.*, 2022).

The use of organic fertilizers derived from manure products and crop residues has gained significant attention in agricultural research due to their potential to promote sustainable and circular farming practices (Kebalo *et al.*, 2024). Organic fertilizer is described as a carbon-rich fertilizer derived from organic materials. These include livestock manure, (vermi) compost, sewage sludge, and other organic materials, which can be applied to soils to supply additional nutrients (FAO, 2019). As the demand for food continues to increase, there is a growing need for innovative and resource-efficient approaches to enhance crop growth and yields. In this context, the development of organic fertilizers presents a promising solution to improve soil fertility, optimize plant productivity and reduce reliance on synthetic fertilizers. Combined with leguminous crops, the application of microbial biofertilizers can enhance soil conditions and N fixation (see Chapter 2).

Crop residues, manure, agro-industrial by-products, and the organic fraction of municipal solid waste are valuable sources of plant nutrients, presenting a potential avenue for enhancing resource utilization efficiency and strengthening the sustainability of agroecosystems. Burning of bio-waste materials not only poses environmental degradation risks but results in the significant loss of essential nutrients inherent in these materials. Key elements such as C, N, P and K and other nutrients, existing in diverse proportions within distinct residues, experience a depletion ranging from 20 to 100 percent through combustion (Kumar *et al.*, 2023). These residues can be utilized through various innovative methods to harness their full potential. Common opportunities for bio-waste utilization include, among others, incorporation into bio-energy generation, organic fertilizers (e.g. composting and biochar production), and fostering mushroom cultivation (Shinde *et al.*, 2022). See section 5.7.2 for a case study presenting the potential to re-utilize FLW in the agrifood system.

Although converting organic residues to fertilizing products is not a novel technology, a composting process still represents one of the most efficient approaches for recycling bio-organic waste into environmentally friendly soil enhancers and plant nutrition products. This controlled aerobic process involves a diverse array of microorganisms that break down various biodegradable organic compounds, converting them into organic fertilizer, thus completing the nutrient cycle. As a biological process, microorganisms are the main drivers behind the composting process (Greff *et al.*, 2022). Providing additional effective microbes during composting can foster the biodegradation process by promoting the diversity and activity of beneficial microorganisms. This, in turn, shortens the production time and enhances the quality of produced compost.

Even though the benefits of composting are evident, the composting process involves N transformations such as ammonification, nitrification, denitrification and $NH_3$ assimilation through microbial community, which inevitably leads to N loss, causing secondary pollution and a reduction in compost quality (Sun *et al.*, 2022). The extent of N loss during composting is closely related to the initial N and C content of the feedstocks. Organic feedstocks contain various N content and forms. Crop residue and garden waste predominantly consist of cellulose and lignin, featuring low N levels. In contrast, sewage sludge and food waste are abundant in N, while manure contains elevated levels of uric acid and ammonia salts, resulting in a high N content with a low C:N ratio. Consequently, adoptable composting strategies are essential to manage diverse organic residues with varying N content (Chen *et al.*, 2023).

Organic fertilizers are considered slow-release fertilizers. Through their production process, they exhibit higher stability and thus have a good slow-release effect during soil application. They provide nutrients in lower amounts over an extensive period aligning with their mineralization rate in soil post-application, facilitated by the active involvement of soil microorganisms. Solid manure- and plant residues-based fertilizers release N slowly, consequently reducing N losses. Although the theoretical expectation is of low NUE when organic-N is applied, Zhu *et al.* (2023) cited high NUE in organically fertilized croplands through field experiments. This improved NUE is attributed to the enhanced organic matter content, as well as the presence of macro-, meso- and micronutrients in organic fertilizers, which promotes soil fertility and enhances crop productivity. Application of organic fertilizers can reduce N fertilizer use by 30–40 percent while maintaining crop yield and increasing soil fertility (Gao *et al.*, 2024; Zhang *et al.*, 2016). Additionally, incorporating organic fertilizers with mineral fertilizers improved soil quality and increased wheat and maize yield by 26.4–44.6 percent and 12.5–40.8 percent, respectively, compared to the recommended mineral fertilizer rate (Zhou *et al.*, 2022).

### 5.3.3 Upcycling agricultural residues and industrial by-products

Globally, an estimated 13 percent of the food produced is lost in the supply chain (from harvesting, post-harvest handling, processing, and distribution before retail) (FAO, 2022d); a further 19 percent of food is wasted in retail and consumption (households, food services, etc.) (UNEP, 2024). This food loss and waste could feed 1.26 billion people yearly and has an estimated economic loss of one trillion USD, emitting 8 percent of the global GHG emissions (FAO, 2022a, 2023d; Santagata *et al.*, 2021). Beyond FLW, agricultural residues (such as straw, manure and forest/pruning residues) and industrial by-products (such as wastewater from cheese production) are currently contributing to the pollution problem instead of being valorized into their untapped potential as raw materials for bio-based products. By repurposing these streams, the circular bioeconomy approach reduces competition for land between feed and food production while recycling nutrients back into the system. Through this, N can either be recycled back into the production system or used as feed, organic fertilizer, or other agricultural input. Section 5.7.3 presents a case study outlining the potential to re-utilize rice straw, an agricultural residue often considered a waste. Non-food crop and forestry production are also part of the bioeconomy and can be used as part of integrated systems and natural resources management to reduce competition of food crops for non-food purposes. The following examples highlight how waste and by-products from the livestock and fisheries sector can be upcycled.

- Wastewater in milk production factories is used to extract valuable nutrients, such as whey, to enhance biogas production and its conversion into electricity.
- Eggshell and eggshell membranes are used for bio-based products (replacing limestone in cement) and services (remediation of acidic soils, and heavy metals removal from soil and water).
- The skin, scales and bones of fish processing are used by the textile and cosmetic industries. Fish scales find applications in manufacturing products such as nail polish or eye lenses. Collagens and gelatine, chitin, fatty acids (particularly polyunsaturated fatty acids such as omega-3 and omega-6), peptides, carotenoids, and minerals are often used as food supplements since they have various biochemical and pharmaceutical applications.
- Plant biomass from grasslands is used for chemicals, energy and fibres.
- Forages based on crop residues that are not suitable for human consumption are used for animal feed and bedding.
- Plasma treatment of organic material, such as dairy manure digestate, is a new method for reducing $NH_3$ and GHG emissions by fixing and stabilizing the N content in the organic material.

Reducing FLW can contribute positively towards multiple SDGs (Mak *et al.*, 2020), but the differences in its composition across countries in terms of embedded nutrients, availability, and environmental impacts are not well known. It is crucial to build the infrastructure and waste collection value chains to allow the upcycling of unavoidable waste into high-value products or services and reduce the uncertainty in the constant biomass availability of supply and the quality of the raw material for industry. It is crucial to improve the planet's ability to feed the growing global population. For instance, reducing 30 percent of food waste would save around 40 million hectares of cropland, and reducing it by 50 percent (SDG goal of halving consumer food waste) presents an opportunity to save roughly USD 380 billion in 2030, given projected growth in food demand and waste (WEF, 2020).

Qualitative and quantitative assessment of the availability of secondary (waste-derived) feedstocks and classifying the different conversion systems into new products is crucial for the transition to happen. The substitution potential is one issue that should be carefully assessed. Also, the economic, social and environmental sustainability of the new products should be properly monitored and evaluated through the overall life cycle sustainability assessment. A particularly useful tool is the cascading approach, which allows assessing the best possible uses for each raw material that provides the highest value first and the lowest one last. The highest value considers not only economic value of the product but also if there is available technology, social and environmental valuation, or access to markets. For instance, N availability in waste varies a lot, and the economic efficiency of its extraction may be less than extracting phosphorous, therefore, by the cascading approach, actors would not consider the option of N extraction.

Regulations should ensure One Health issues, such as safety for humans, animals and the environment, for the effective implementation of circular bioeconomy practices. An example is the use of insects fed with urban waste to produce flour for feed meals to reduce the possibility of transmission of agricultural chemicals, antimicrobial resistance, etc. Innovations such as stable isotopes and molecular/genomic techniques offer approaches to detect potential antimicrobial resistance in farm animal environments (water and soil) and ensure early action in pollution reduction.

Many agricultural practices exist to increase NUE and maximize the amount of N that cycles through the agricultural system (see Chapters 2 and 3). Inevitable losses occur during different steps of the agricultural production chain, including during harvest, excretion and storage of manure, application of organic and synthetic fertilizers, and losses during post-harvest and food processing stages.

The anaerobic digestion of manure and crop residues or waste streams from the processing industry is a widespread technology to convert biomass into bioenergy. During the breakdown of organic matter, biomethane is formed, which can be converted to renewable fuels (such as gas, electricity and heat). The by-product from anaerobic digestion is called digestate, which can be used as organic fertilizer. Through this, N can be used in the production system. At the same time, substantial reductions of GHG (in particular $CH_4$) from manure can be achieved, as biogas is captured in the biodigester rather than emitted to the atmosphere while producing renewable energy. Various studies have shown that significant GHG reductions can be achieved when the slurry is treated through biodigestion (Burg *et al.*, 2018; Møller *et al.*, 2022; Wattiaux *et al.*, 2019). Digestate is characterized by a low organic matter content and a high ammoniacal N content. This increases crop N availability but increases the risk of $NH_3$ emissions during storage and digestate application. To minimize N losses and increase the recycling of N in the system, covered storage or further treatment of digestate is necessary. An overview of different treatment options is outlined in Chapter 3, as well as more details on the benefits of anaerobic digestion for both (semi)industrial and smallholder livestock farms.

## 5.4 PROMOTING CIRCULAR BIOECONOMY APPROACHES IN LIVESTOCK SYSTEMS TO SUPPORT BETTER NITROGEN MANAGEMENT

Intensive livestock systems are characterized by the dependence on the import of livestock feed, and as a result, local livestock populations in livestock-dense areas can grow beyond the national feed capacity. This results in a decoupling of local livestock and feed production, which is an important cause of a disruption of the N cycle, leading to an accumulation of resources in livestock-dense areas through manure production and a depletion of resources in regions where biomass is exported through animal feed trade (Wang *et al.*, 2022b). This results in an increase in synthetic fertilizer use for feed production. The detrimental effects of excessive losses of N to the environment have been described in Chapter 4, as well as the significant role livestock production plays therein (Chapter 3). More recently, the recoupling of local livestock and feed production has been shown to solve multiple challenges linked to livestock production, such as decreasing GHG and $NH_3$ emissions, minimizing leaching of N to the environment, and enhancing biodiversity in agricultural areas (Billen *et al.*, 2021, 2024; van Selm *et al.*, 2023; Zhang *et al.*, 2019a). In order to recouple local livestock and feed production, the numbers of livestock numbers kept in a region is dependent on the feed-producing capacity of that region. In this way, local N cycling is more balanced, and losses are minimized, which should be combined with maximizing the recycling of N through manure, contributing to circular bioeconomy principles (van Zanten, van Ittersum and de Boer, 2019). A recent study by van Selm *et al*. (2023) calculated the impact of recoupling livestock numbers to local feed production for the Kingdom of the Netherlands, a country heavily dependent on imported feed and with ambitious goals to reduce $NH_3$ and GHG emissions by agriculture to tackle the nation's environmental challenges (RIVM, 2022). Results of this study show that with recoupling local livestock and feed production, national targets to reduce agricultural GHG and $NH_3$ emissions can be met and the NUE of the food system increased to 39 percent, compared to 31 percent with the business-as-usual scenario. Furthermore, significant environmental benefits would be present, with a reduction in acidification potential of 48 percent, GHG ($N_2O$ and $CH_4$) emissions would be reduced by 30 percent, and $NH_3$ emissions by 50 percent. A transformation of the livestock sector would be necessary when recoupling livestock-dense areas to local feed production, as a lower livestock number could be supported (van Selm *et al.*, 2023). Similar benefits of recoupling livestock and feed production have been shown by Zhang *et al.* (2019a) in Southeast Asia, where recoupling livestock production is increasingly recognized as a critical measure for sustainable agricultural development. If livestock were to be fed with domestic feed supply only, N losses through emissions, runoff and leaching would be reduced considerably, whereas the amount of N recycled to croplands would more than double compared to a business-as-usual scenario. It is, however, crucial that manure management systems are improved to further reduce N losses to the environment and increase overall NUE. Furthermore, recoupling crop and livestock systems can pose geographical challenges as well, especially in regions where livestock production is concentrated in certain areas. This is the case in Southeast Asia, where a recent study shows that reintegration of livestock and crops would mean a relocation of five billion animals away from livestock-dense areas (Bai *et al.*, 2022). As a result, N losses and $NH_3$ emissions could decrease by 77 percent and 63 percent, respectively, and fertilizer use would decrease by 82 percent if manure were properly managed and used as organic fertilizer. Rethinking local feed production for livestock production in livestock-dense areas where environmental pollution of N is prevalent can contribute to the transformation of agricultural systems and the development of a circular bioeconomy while significantly reducing N pollution.

When enhancing circular bioeconomy would be further developed by increasing the resource use efficiency and circularity in agrifood systems, the opportunities to minimize feed–food competition, increase the recycling of waste streams for feed, and minimize food losses can be explored. Recent studies have shown that the implementation of a zero feed–food competition scenario would result in a transformation of the food production system where livestock is solely fed from marginal lands and by-products, and croplands are used to produce crops for direct human consumption (van Kernebeek *et al.*, 2016; van Selm *et al.*, 2023; van Zanten, van Ittersum and de Boer, 2019). As aforementioned, two-thirds of current agricultural land is considered marginal, and livestock plays a key role in converting this biomass to high-quality food for human consumption (Mottet *et al.*, 2017). The remaining agricultural land suitable for crop production would be used primarily to produce crops for direct human consumption to maximize resource use efficiency, including NUE, on these lands. In this scenario, the number of livestock kept in livestock-dense areas would decrease. Still, the amount of crops cultivated for direct human consumption would increase and thereby contribute to an increase of available food, which in turn can increase global food security (van Kernebeek *et al.*, 2016). Different studies, as summarized by Oosting *et al*. (2022), indicated that circular food systems can provide up to 36 grams per capita of animal-derived protein per day. At present, in high-income countries, animal-derived protein consumption is close to 60 grams per capita per day. Implementation of such a system would thus mean a shift to a more plant-based diet for countries where the amount of TASF exceeds the recommended quantities (Beal *et al.*, 2023; Billen *et al.*, 2024; Leip, Bodirsky and Kugelberg, 2021; Sutton *et al.*, 2013). This approach would not only contribute to healthy diets, as over-consumption of animal-derived proteins is linked to multiple health concerns and diet recommendations of TASF are exceeded in many wealthy countries (Chatzimpiros and Harchaoui, 2023; Willett *et al.*, 2019), but also benefit the sustainability and efficiency of agrifood systems in terms of natural resource use. Simon *et al*. (2024) calculated that a shift from over-consumption to recommended protein intake levels in the European Union, combined with integration of circularity principles, would reduce land use by 58 percent and decrease GHG emissions by 80 percent. Decreasing TASF consumption in high-income countries is proposed by multiple studies as a win–win solution for the sustainable development of agrifood systems and transition to healthier diets (Billen *et al.*, 2021, 2024; Erisman *et al.*, 2018; Sutton *et al.*, 2013), resulting in a reduction of N losses to the environment and increased food production efficiency and food security. On the other hand, the development of circular agrifood systems in LMICs provides an opportunity to increase NUE and the sustainable development of livestock systems (Oosting *et al.*, 2022). Furthermore, they can provide increased availability of TASF in regions of under- and malnutrition where increased consumption of TASF is seen as a key strategy to address food security and nutrition (Beal *et al.*, 2023; Oosting *et al.*, 2022). Transitioning to agrifood systems in a circular bioeconomy in LMICs can enhance the sustainable development of livestock systems and boost food availability and security.

A study by Billen *et al*. (2015) explored the implications for the global agrifood system when meeting the requirements to feed the projected world population equitably. Here, an equitable diet is defined as a diet that meets all dietary requirements and can be shared by all regions of the world, hence providing food security for all (which directly supports SDG 2 (UN, undated)). The exercise by Billen *et al*. (2015) showed that an annual diet containing 4 kg N/capita with 40 percent animal protein or 5 kg N/capita with 20 percent animal protein would be feasible as an equitable diet for the growing world population, meaning food security is met for all regions of the world. Furthermore, these diets are in line with the per capita protein intake recommendations of the World Health Organization (Joint WHO/FAO/UNU Expert Consultation, 2007). Besides an equitable distribution of available nutrients, this global agrifood system would require shorter supply chains and cause less agricultural N pollution globally through an increase in NUE of both crop and livestock systems (Billen, Lassaletta and Garnier, 2015).

### 5.4.1 In focus: Circular bioeconomy for innovative animal feed production

Animal feed production from food waste has caught the attention of different countries. Six bioeconomy strategies include specific actions and targets on feed, often using waste (FAO, 2024b). For example:

- The use of by-products from biofuel production and development of new sources of protein (e.g. insects) can reduce feed imports.
- Development of biological fodder such as enzyme formulations, microbial agents, fermented feeds, and feeding amino acids to address major problems in farming such as feed safety, scarcity of raw materials and environmental pollution.
- Knowledge, technology and regulations should be developed to produce food and feed ingredients from local fruit, vegetable and meat industry by-products.
- Increased use of Norwegian ingredients in the production of feed when it is profitable and environmentally sustainable.

A recent FAO report states that the livestock sector can change the sources of its feed to promote innovative solutions for reducing its environmental footprint (FAO, 2023d). A systems-wide change is needed to find the best technologies and solutions for upcycling waste, residues and by-products into novel feeds and reduce the amount of land required for growing feed crops, reducing GHG emissions, and restoring polluted ecosystems.

Alternative feeds from by-products and waste that replace conventional livestock feed can increase NUE through an increased efficiency of resource use. As conventional livestock feed is associated with high N losses, especially through the application of synthetic fertilizer, a decreased use of this type of feed can contribute to increasing NUE of the livestock production chain. Upcycling of waste, residues and by-products contributes to closing nutrient loops, including N, of the agricultural production system, thereby minimizing N losses. While fibrous lignocellulosic materials can be used for ruminant diets, animal-based co-products (meat and bone meal) can improve the protein meals of pigs, poultry and aquaculture. These different solutions in the circular bioeconomy can make feed more affordable and sustainable, lessening the demand for traditional feed materials, which are resource and environmentally intensive and contribute to feed–food competition (Nath *et al.*, 2023). The use of organic waste should ensure that feed safety is not compromised. Modern technologies can help to improve the quality of organic waste as feed and be incorporated into animal diets. Rajeh *et al*. (2021) studied different types of FLW and different substitution rates to see how animal growth performance varied and found that there was no difference in animal growth or health using partial incorporation of food waste in the animal diet.

The solutions described in this chapter are a selection of measures and innovations that could enhance the circular bioeconomy and sustainable use of N. The effectiveness with which this is done must be measured and quantified to confirm its contribution to increased NUE in agrifood systems. The development of metrics for impact assessment is important and often a limiting factor in determining if technologies are suitable for specific (local) conditions or contributing to specific goals. For example, a study by Lavallais and Dunn (2023) found that several specific technologies to recover N from pig manure did not yield a substantial increase in nutrient circularity and recovery. This shows that the implementation of specific technologies might not always increase N recovery or NUE. Metrics are a valuable tool to assess the impact of the suitability of technologies.

## 5.5 FAO'S WORK ON BIOECONOMY

FAO has elevated the concept of sustainable bioeconomy to its Strategic Framework 2022–2031 and the work of its Governing Bodies. As part of its Programme Priority Area "Bioeconomy for Sustainable Food and Agriculture", FAO emphasizes sustainable and regenerative production designed to reduce reliance on harmful chemicals, enhance circular practices, conserve biodiversity and ecosystem services, and enhance climate resilience. FAO highlights "Healthy People, Healthy Planet" principles across areas of practice, including sustainable and regenerative bioeconomy alternatives and bio-innovations designed to promote net zero, climate-resilient, nature-positive, zero waste and pollution-free agrifood systems.

FAO is engaged with relevant international conventions such as the Stockholm Convention, Basel Convention, and the Strategic Approach to International Chemicals Management and leads United Nations work on sound management of pesticide life cycles. Together with the World Health Organization, FAO developed the International Code of Conduct on Pesticide Management and assists Rotterdam Convention Parties in regulating hazardous pesticide formulations. FAO leads the Global Alliance on Highly Hazardous Pesticides endorsed by the Strategic Approach to International Chemicals Management during the Fifth Meeting of the International Conference on Chemicals Management, and has published the International Code of Conduct for the Sustainable Use and Management of Fertilizers. It has produced the Voluntary Code of Conduct for Food Loss and Waste Reduction, leads the United Nations activity on soil plant nutrition and sustainable soil management and is leading the development of the Voluntary Code of Conduct for sustainable use of plastics in agriculture. This work contributes to the Global Framework on Chemicals – For a Planet Free of Harm from Chemicals and Waste.

FAO's work on bioeconomy strategies around the world shows how countries are tackling agrifood systems in their bioeconomy strategies and how bioeconomy practices are recognized as contributing to climate, biodiversity, food security and nutrition goals. Currently, 21 countries and three regions have implemented specific bioeconomy strategies. Additionally, approximately 35 countries have strategies related to bioscience and biotechnology relevant to the agrifood sector. This coverage is expanding rapidly; FAO is tracking the development of 17 more dedicated bioeconomy strategies in progress in an online dashboard. Countries explicitly include bioeconomy in their other programmatic documents on climate, biodiversity and food security strategies. For instance, countries often explore climate change benefits and trade-offs in the development of their bioeconomies (Table 5). As of November 2024, ten countries explicitly included bioeconomy in their Food Systems Transformation Pathways, and most of the 127 submitted Pathways prioritized related approaches such as sustainable consumption and production.

FAO's programme on sustainable bioeconomy approaches offers a range of solutions, from biofertilizers such as microbial fertilizers to bio-based solutions for soil remediation, reuse of wastewater for agriculture, anaerobic digestion, and upcycling of organic waste to convert it into a resource and avoid associated emissions (including crop residues and livestock manure).

FAO supports countries and advocates for avoiding chemical use in agrifood systems (including agro-industries) and restoring polluted sites. On the latter, bioremediation is an example of implementing bioeconomy practices to treat soil or water resources affected by excess input of nutrients. Bioremediation uses microbes that remove contaminants

TABLE 5
**Main climate change trade-offs (–) and synergies (+) between bioproducts and climate change**

| Stages of the bio-based economy value chain | GHG emission reduction | Sequestration | Climate change adaptation |
|---|---|---|---|
| **Overall** | + Most bio-based products have a lower GHG footprint compared to fossil products | + Bioproducts sequester $CO_2$ during their lifetime | + Higher environmental and livelihood resilience |
| **Biomass production** | – Production of biomass can increase GHG emissions<br>+ Biomass production can be optimized by climate-smart practices | + Carbon sequestration in agricultural soils (if good soil and water management practices), forests and oceans | + Higher environmental resilience if natural resources are sustainably managed<br>– Climate change impacts can reduce production of local bioproducts and force production to new locations |
| **Bioproduct processing** | + Most bio-based fuels, chemicals and polymers have lower GHG emissions compared to petrochemical products<br>– The manufacturing of bioproducts uses significant amounts of fossil energy anyway<br>+ New biotech pathways can improve energy consumption<br>+ Local production reduces GHG emissions from transport | + Future carbon capture and use technologies will use renewable $CO_2$ sources | + Localized production increases employment opportunities and improves rural economies |
| **Use phase (cascading)** | + Circular long-lasting bioproducts show a reduction in GHG emissions<br>– Recycling potentially adds to total energy consumption and GHG emissions | + Long-lasting products can sequester carbon over the long term<br>+ $CO_2$ sequestration can be increased through the cascading use of biomass | + The use of traditional, local bioproducts (construction materials, medicine, energy) has specific benefits |
| **End of life** | + Incineration substitutes fossil energy<br>+ / – Only in certain applications is biodegradation a viable option | | |

*Source:* **Gomez San Juan, M., Bogdanski, A. & Dubois, O.** 2019. *Towards sustainable bioeconomy: Lessons learned from case studies*. Rome, FAO.

such as oil, solvents and pesticides. Microorganisms are an integral part of the bioeconomy and can greatly contribute to the challenges associated with N use in agriculture. The use of efficient microorganisms as growth promoters is a soil fertilization alternative in many countries and biophysical conditions (Gomez San Juan, Harnett and Albinelli, 2022b).

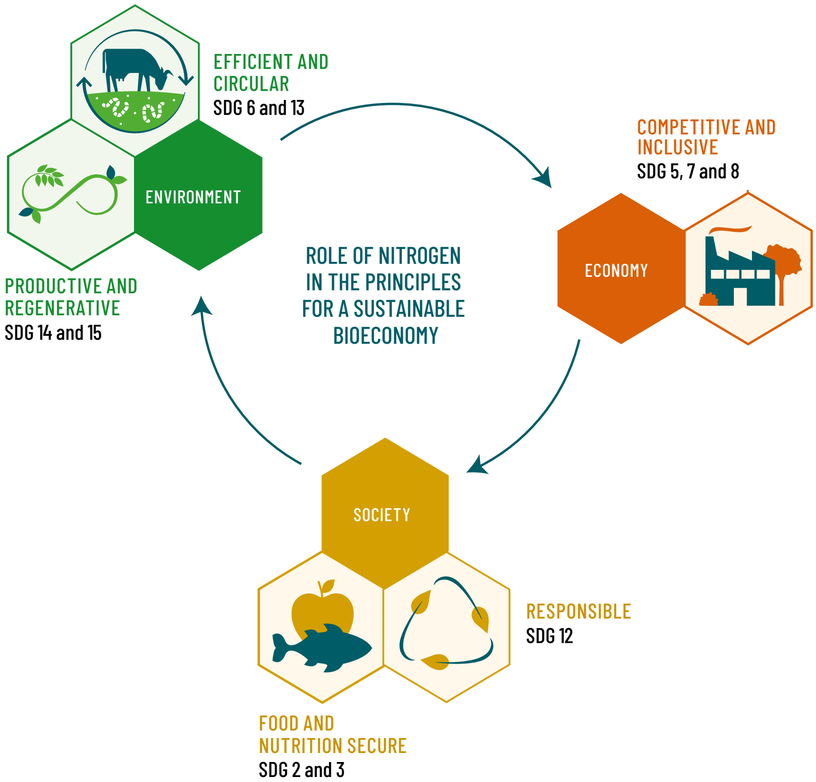
Generally, when governments or regional/local authorities develop strategies and policies to support a sustainable bioeconomy, they seek to improve the production, use, consumption and conservation of biological resources. Often, enhancing circularity increases efficiency in using and regenerating these biological resources, thus reducing pressure and competition between the end-use sectors. For instance, crop residues can be left on the soil to enhance soil structure and nutrient content and/or used for feed or as a substrate to grow food such as mushrooms and/or in pulp and paper, construction, chemicals, energy, or textile sectors.

These bioeconomy strategies or policies lay the groundwork for bio-based research and new technology development in agrifood systems, which can further enhance efficiency and competitiveness. This is often used in the cascading approach to prioritize biomass end uses with higher added value. FAO supports countries and regions in designing, implementing and monitoring bioeconomy strategies and programmes using the Aspirational Principles and Criteria for a Sustainable Bioeconomy in a framework called "FAO Bioeconomy Toolbox". Existing bioeconomy practices often lack an integrated consideration of social, economic, environmental and governance goals and trade-offs between them. FAO and its partners have developed a set of 10 Aspirational Principles and 24 Criteria aimed at facilitating such a holistic approach (FAO, 2021). Figure 31 shows which Aspirational Principles link to nitrogen and how sustainable nitrogen management can contribute to these Aspirational Principles.

## 5.6 CIRCULAR BIOECONOMY IN THE CONTEXT OF FOOD SAFETY

The transformation of agrifood systems towards circularity needs to go hand-in-hand with the adaptation of policies and risk assessments to ensure food safety. As those assessments stem from processes developed for linear systems, new data needs to be generated to fill gaps in possible risks related to circular agrifood systems. FAO analysed food safety in a circular economy context, focusing on four areas: water recycling and reuse, integrated farming systems, food waste and by-products, and food packaging

FIGURE 31
**The role of N in the FAO Aspirational Principles and Criteria for a Sustainable Bioeconomy and the SDGs**




*Source:* Authors' elaboration based on **FAO.** 2021. *Aspirational principles and criteria for a sustainable bioeconomy.* Rome, FAO.

waste (FAO, 2024b). In these areas, the focus should be on monitoring contaminants in recycled water, transmission risks of pests and pathogens in integrated systems with multiple livestock species, contaminants in food waste used as feed, and potential food safety risks linked to reusable and recycled packaging.

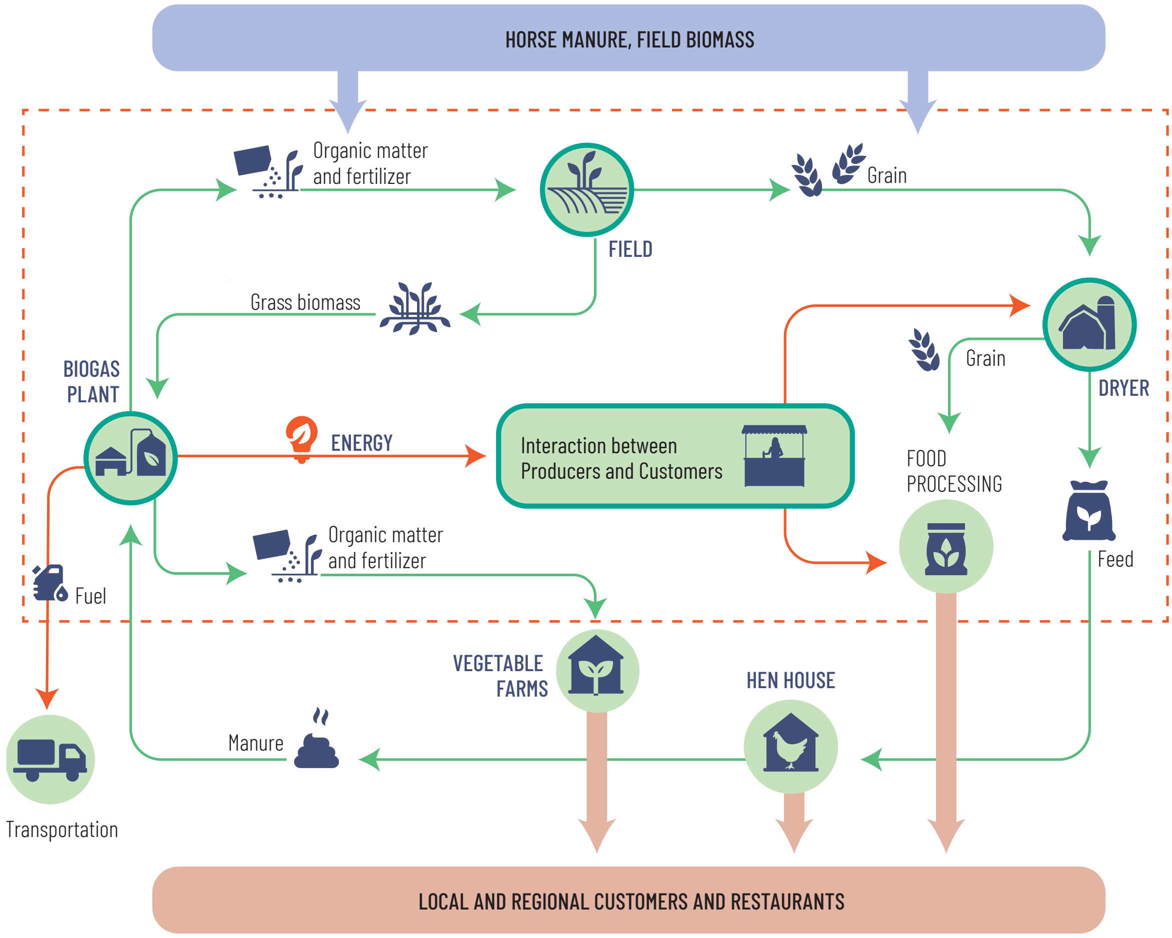
## 5.7 CASE STUDIES

### 5.7.1 Exchange of nutrients across farm elements – An example from Finland

Enhancing circularity on farm or between farms can increase NUE, sustainable management of natural resources, and decrease the negative impacts of agricultural activities on the environment. An example of a circular farming system, where multiple farms and farm elements work together, can be found in Palopuro village, Finland. The organic farming system, with 385 ha of farmland, is comprised of an arable farm (Knehtilä farm), a vegetable farm (Lehtokumpu farm), a poultry farm (Mäntymäki hennery), and a biogas plant (Figure 32). As an organic farm, farmland is fertilized through green manure leys. The nutrient use efficiency in the use of green manure leys is challenging as the timing of N mineralization, and thus N availability to plants, does not meet the peak demand of crop plants, and ploughing green manure is linked with N losses in certain seasons. To increase NUE, the farming systems used biomasses not competing with food production as input for their biogas plants, such as green manure, straw, chicken manure and horse manure from nearby stables. The digestate is used as fertilizer, and the energy produced is either used for food processing activities on farm or upgraded to biomethane to be used as fuel.

The effects of the above farming practices have been analysed by Koppelmäki *et al*. (2019) and have shown that multiple benefits can be seen from this integrated system. Using digestate instead of green manure as fertilizer allows for better nutrient utilization, as digestate can be applied on any field where it is most necessary as compared to green manure, which is applied on the same field in which it was grown. Furthermore, an increase in mobile N input and a decrease in N losses resulted in an overall reduction of N surplus of 38 percent. Overall, NUE of the farm increased,

FIGURE 32
**Overview of the circular farming system in Palopuro village, Finland**




*Note:* Green arrows indicate the flows of biomass, and the red arrows indicate the production of energy and fuel.

*Source:* Authors' elaboration based on **Koppelmäki, K., Parviainen, T., Virkkunen, E., Winquist, E., Schulte, R.P. & Helenius, J.** 2019. Ecological intensification by integrating biogas production into nutrient cycling: Modeling the case of Agroecological Symbiosis. *Agricultural Systems*, 170: 39–48. https://doi.org/10.1016/j.agsy.2018.12.007

and the projected cereal yield through increased quantity of available N to crops increased by 40 percent. Lastly, through the production of biogas from biomass, the farming system became a net energy producer, producing 70 percent more biogas than needed for on-farm operations.

This farming system shows how different elements within a farming system can be connected and, through this, enhance NUE. Implementing such a system on farms or regions different from the one described above requires assessing what measures are effective for specific conditions. Furthermore, farming systems can be further optimized by determining which combinations of elements within the system create the most benefits. Koppelmäki *et al*. (2021) analysed different optimization scenarios for the Palopuro farm in Finland to determine which system would produce the most benefits compared to the current scenario, as described above. These included (1) a scenario where 20 percent of the area designated for annual crops was used to produce a clover–grass mixture to increase biological N fixation and produce biomass for biogas production and (2) the inclusion of dairy cows, with the herd size determined by on-farm silage production where 20 percent of cropping area was used for this purpose. The study showed that with the first scenario, energy production could be almost doubled compared to the current scenario, with a reduction of human digestible protein production of 13 percent. Introducing dairy cattle in the second scenario increased human digestible protein production by 164 percent, while the externalities were reduced as feed imports from the system were reduced. The energy production was reduced by 8 percent compared to the previous scenario. Both scenarios enhanced N recycling substantially. This analysis shows that the introduction of different farm elements has different effects on the farm outputs, and decisions on what type of management practices to include depend on priorities and focus of farm activities and preferred outputs.

### 5.7.2 Circular bioeconomy in Abidjan: from food waste to fork

FAO is implementing a project in Abidjan where black soldier fly is reared on urban waste and used for chicken feed and frass (a by-product of raising insects) is used for fertilizer (FAO, 2024b). Local partners (The Institute of Circular Economy of Abidjan and the youth startup Bioani) have built a 1000 $m^2$ black soldier fly farm in Abubo. Every day, around 1000 kg of organic waste is collected from local markets and brought to farms, where organic waste is converted by black soldier fly larvae into proteins for poultry feed. Moreover, the frass is used to produce organic fertilizer. Around 120 kg of final dried larvae product for feed are produced, as well as 200 kg of organic fertilizer from the frass, which are sold to urban and peri-urban farmers at a cheaper price than synthetic fertilizer. The farm currently employs ten young people and women and serves as a farm field school, where ten more people are being trained. Nitrogen is an important nutrient for plant growth and essential, along with P, for protein formation. Some studies analysed the composition of nutrients, such as N for protein intake, and novel bioactive compounds (Prandi *et al.*, 2019; Pyett *et al.*, 2023). For instance, a study examined the efficacy of recycled food-waste-based feed for laying hen performance egg quality, and nutrient digestibility. Dao *et al*. (2023) compared feed based on wheat, sorghum, and soybean meal, a recycled food-waste-based feed, and a 50:50 blend of the two. Hens offered food-waste-based diets that had similar egg weight, hen day egg production, and egg mass but lower feed intake and higher feed efficiency compared to those fed with wheat, sorghum and soybean meal. Hens fed the food waste diet had lower shell-breaking strength and shell thickness, higher yolk colour score, and higher fat digestibility. Feeding the recycled food-waste-based feed maintained egg production while improving feed efficiency.

### 5.7.3 Rice straw utilization in the circular bioeconomy

Rice is a crucial food crop representing the major food staple for more than half of the world's population in Asia, Latin America and parts of Africa (Fukagawa and Ziska, 2019). Over 158 million hectares worldwide are cultivated with rice, resulting in approximately 700 million tonnes of rice straw being left behind annually (Bhattacharyya *et al.*, 2021).

Rice straw is commonly burned in the field, releasing significant GHG emissions, including 0.7–4.51 g $CH_4$/kg and 0.019–0.069 g $N_2O$/kg of burned straw. This process contributes to the dispersion of air pollutants and the depletion of N and organic matter in the topsoil (van Hung *et al.*, 2020). Management of such a large quantity of rice residues in a sustainable manner poses a considerable challenge and requires innovative strategies and approaches. This task becomes even more critical given the limited time frame of 10–15 days available between rice harvesting and sowing the next crop as highlighted by Thakur *et al*. (2018). Promoting rice straw utilization through circular bioeconomy approaches in agrifood systems addresses optimization and enhances natural resource use efficiency while minimizing environmental pollution. Circular bioeconomy principles transform rice straw from waste into a valuable resource, with strategies including composting for soil fertility, renewable energy generation, and value-added product creation. These approaches improve environmental resilience in agrifood systems.

An example of rice straw utilization through circular bioeconomy principles can be seen in India, where rice straw burning is a major problem. With only about a 20-day window for farmers to clear fields of rice straw during the harvesting period, which they have little ability or capacity to collect, bail and store, burning rice straw is the common practice. This contributes to an increase in air pollution. In Punjab, a value chain has been developed that enables rice farmers to sell a percentage of their straw for the production of compressed biogas and biomass pellets. This generates an additional source of income and reduces burning incidents and air pollution, as well as being a substitution for fossil fuels.

A model crop residue value chain is illustrated by FAO (2022b) to facilitate the harvest, collection, transit and storage of the rice straw. It shows the investment needed to supply the rice straw. The study provides a techno-economic analysis of energy technologies that drive rice straw's potential use and profitability for sustainable energy alternatives. Results suggest that 30 percent rice straw can provide all the required Punjab Compressed Biogas while using 15 percent of this energy for biomass pellets and reducing coal use by 72 percent. This application would encourage entrepreneurship at the local level, raise farmers' incomes, and decrease open burning of rice straw, air pollution and climate change.

## 5.8 CONCLUSION AND KEY MESSAGES

Agrifood systems in a circular bioeconomy can support sustainable N management through the implementation of circular principles that minimize N emissions and pollution. Circular agricultural systems focus on maximizing the efficiency with which food is produced by maximizing resource use efficiency and minimizing loss. This is achieved by (1) avoiding FLW, (2) recycling unavoidable FLW back into the agricultural system, (3) using available cropland to produce crops for direct human consumption, and (4) using livestock to convert biomass unsuitable for human consumption.

Upcycling of by-products from the agrifood system can reutilize N that would otherwise be lost to the environment. Converting these by-products to organic fertilizers ensures the recycling of N and can substitute synthetic fertilizers without compromising the productivity and yields of crop systems. In a circular agrifood system, manure is not seen as a waste product but a valuable resource of nutrients that forms a source of N for crop production. Manure can either be used as organic fertilizer or processed via anaerobic digestion to produce both bioenergy and digestate, with the latter returned to the cropping system as organic fertilizer.

For livestock, recoupling of local feed production and livestock can increase NUE of livestock systems substantially and result in a significant decrease in N emissions and pollution. Pig and poultry systems can recycle N by using FLW and by-products from the food industry as feed, thereby increasing NUE, minimizing N loss, and avoiding feed–food competition. Incorporating these circular principles may result in a decreased number of livestock that can be supported in countries with a high dependency on imported concentrated feed, which would necessitate a shift to a more plant-based diet in countries where TASF overconsumption occurs.

Policy can support the transition to circular agrifood systems by incentivizing the adoption of circular practices, supporting farmers in the transition to circular farming systems and promoting the incorporation of a more plant-based diet for consumers.

# Chapter 6
# Policy instruments to promote sustainable nitrogen management

Extensive research has contributed to the understanding of the N cycle and the role of reactive N as an essential agricultural input and a critical environmental pollutant of the aquatic, terrestrial and atmospheric environments (Galloway *et al.*, 2003, 2013). The number of policies that influence N use and management in agrifood systems has been rising, driven by the fact that these systems represent the largest global source of N pollution. This is mainly due to high N input coupled with low NUE (Cassman and Dobermann, 2022). These policies range from voluntary adoption of N management practices by farmers to government policies and regulations. They operate at multiple levels, from local to global, and can be divided into policies incentivizing N use through synthetic fertilizer and manure and policies targeting N pollution in the environment. Most policies in agrifood systems incentivize N use, which reflects the primacy of food security over environmental concerns (Kanter *et al.*, 2020a).

## 6.1 NITROGEN IN CROP PRODUCTION POLICIES

### 6.1.1 Fertilizer access, agriculture policies and nitrogen use efficiency

Fertilizers and other input subsidies are popular policy interventions in which governments provide financial support for agricultural development, often aiming to increase crop yields and farmers' incomes, while reducing hunger and poverty (Zhang *et al.*, 2021). Providing synthetic fertilizers and other agricultural inputs and technologies, such as expansion of irrigated areas, high-yielding variety seeds, pesticides, and machinery, to farmers has contributed to the Green Revolution in Asia (Hazell, 2009; Tewatia and Chanda, 2017). For example, wheat production in Asia increased from 46 million tonnes in 1961 to 343 million tonnes in 2022 (FAO, 2024c). The crop yield increased in South Asia (Tewatia and Chanda, 2017) and Southeast Asia (Huang *et al.*, 2017) due to their fertilizer policies and crop genetic improvement. Conversely, despite governments' input subsidy programmes and their associated reforms (the so-called "smart subsidies"), an African Green Revolution did not materialize (van Ittersum *et al.*, 2016). The African Union has established the Comprehensive Africa Agriculture Development Programme to eliminate hunger and reduce poverty by investing in agricultural development to double productivity through access to inputs, irrigation technology, and mechanization (African Union and AUDA-NEPAD, 2024). Van Ittersum *et al*. (2016) estimated that cereal self-sufficiency would decrease by 33 percent in Western Africa and 48 percent in Eastern Africa by 2050 due to climate change, lack of access to inputs, and an increasing human population. In Southern Africa, small improvements in maize yields have been recorded, but these yields are still far below the potential of more than 3 tonnes per ha. While crop production followed opposing trends in Asia and Africa, NUE showed similar trajectories of being consistently low (van Ittersum *et al.*, 2016). Farmers in Asia have applied large amounts of N per ha to meet crop N needs. In contrast, in Africa, the use of synthetic fertilizer has slightly declined because of high fertilizer prices, weak supply, limited investment, and poor and variable crop responses to fertilizer (Nziguheba, van Heerwaarden and Vanlauwe, 2021; Sileshi *et al.*, 2022). Technological innovation and crop genetic improvement programmes, supported by sufficient access to fertilizer, have continued to increase crop productivity in Europe, North America, Latin America and the Caribbean. The following section provides an overview of agricultural policies and impacts on NUE in Asia, Africa, the European Union, North America, Latin America and the Caribbean.

#### *Overview of agricultural policies and impacts on nitrogen use efficiency in Asia*

Asia is the largest consumer of synthetic N fertilizer and provides food, feed and fibre to 59 percent of the world's population (FAO, 2024c). For instance, government policies in China have promoted synthetic fertilizer use and have driven fertilizer overuse and over-application since the 1970s (Ju *et al.*, 2016; van Wesenbeeck *et al.*, 2021). Historically, after transitioning from the People's Commune System to the Housing Responsibility System in 1978, N input per ha increased, resulting in high crop production surpassing household demand (Zhang, 2011). As a result, the food market developed due to the removal of on-farm taxes, further incentivizing farmers to increase agricultural production and apply yet more N fertilizer. Furthermore, fertilizer manufacturing subsidies were introduced to provide cheap use of electricity, natural gas and transport, including preferential taxation policies for fertilizer manufacturers. Fertilizer manufacturing subsidies and other policies were a major driving force for increasing fertilizer availability and accessibility at lower prices (Li *et al.*, 2013).

Unfortunately, the increasing use of N was accompanied by low NUE, resulting in severe environmental externalities. In 2017, about 30 percent of global N fertilizer was applied to Chinese croplands (Yu *et al.*, 2021). In the same year, crop NUE was around 32 percent, which was significantly lower than the world average of 55 percent (MOA, 2019). This situation resulted in agricultural subsidy reform to curb N fertilizer use and reduce environmental externalities (Fan and Yang, 2024). For instance, the fertilizer manufacturing subsidies in China were discontinued from 2015 to 2018 (Wu *et al.*, 2024) and replaced with the "Zero Growth in Synthetic Fertilizer Use" policy by 2020. The latter policy promotes precise fertilization by offering tailored fertilizer recommendations for various regions and crops, adopting enhanced efficiency fertilizers (Kanter and Searchinger, 2018), and substituting synthetic fertilizers with manure (Lin, Xu and Wang, 2022). To this end, policy guides and action plans to speed up the appropriate use of manure were launched (Wei *et al.*, 2021) and included, for example, the "Action Plan for Manure Nutrient Usage (2017–2020)", which aimed to increase the utilization of animal manure in croplands to 75 percent. Given the high level of technological and incentive adjustments required for using manure, the government has offered manure-use subsidies to incentivize this transition (Wang *et al.*, 2023). The focus is on mobilizing this transition since effective use of manure should ultimately pay for itself because fewer synthetic fertilizers are needed. Most recent data shows that manure utilization was 70 percent in 2017 and is targeted to reach 90 percent in 2035 (Wei *et al.*, 2021). As a result of these policies, China's fertilizer use declined by about 7.2 percent (Fan *et al.*, 2023; Ji, Liu and Shi, 2020), which is expected to increase NUE (Sapkota, Bijay-Singh and Takele, 2023).

In India, fertilizer subsidies represent the second-largest expenditure at USD 11.2 billion annually (IMF, 2015). Because of the fiscal burden, India has reduced subsidies on P and K fertilizers. Urea, which is the most widely used fertilizer in the country, remains heavily subsidized, leading to increased N consumption by agriculture. Due to the prevalence of rain-fed agriculture, the application of N fertilizer is often not synchronized with soil moisture availability. Given the prevalence of highly subsidized fertilizers, where farmers only pay a small fraction of the cost, this situation has led to excessive application of N fertilizers per ha in an attempt to maximize crop yield (Bijay-Singh, 2022). As a result, a significant portion of the added N is lost to the environment, resulting in low NUE and air and water pollution risks. Since 2015, all prilled urea has been replaced by neem-coated urea, where neem oil is a natural denitrification inhibitor, which helps improve NUE (Tewatia and Chanda, 2017). The Indian Government is promoting a range of other approaches that are relevant to sustainable N management. For example, Zero Budget Natural Farming avoids the use of synthetic chemical inputs, including N fertilizers, focusing on mobilizing nutrients through stimulated microbial decomposition of organic mulches and by biological N fixation (Smith *et al.*, 2020). The use of "nano urea" is promoted, a proprietary urea solution in water, though its cost-effectiveness remains a matter of debate (Frank and Husted, 2024).

***Overview of current agricultural policies and impacts on nitrogen use efficiency across sub-Saharan Africa***

Despite the introduction of a range of input subsidy programmes (ISPs), including smart subsidies in different countries, crop yields in most African countries remain low. The gap between actual and potential yields, in addition to that between crop N removal and fertilizer N input continues to widen in many countries. This gap causes low fertilizer profitability for farmers, soil nutrient stock depletion, soil acidification, crop yield decline, and even depresses African fertilizer demand because of low crop response (Guèdègbé and Doukkali, 2018). The average level of N fertilizer use is 15 kg/ha, and only a few African countries record fertilizer consumption that exceeds 50 kg/ha (Guèdègbé and Doukkali, 2018). As a result of low fertilizer adoption and application, ISPs have failed to close the yield gap and trigger an African Green Revolution. Jayne *et al*. (2018) attributed the low crop response to fertilizer to several factors, including the limitation of water availability and the widespread presence of low-quality fertilizer in Africa. Many soils are sandy, with low SOM content and low pH, all of which affect nutrient availability. For instance, in Southern Africa, soils have been reported to have reached a tipping point, where SOM is below the minimum threshold to support crop productivity (Messina, Peter and Snapp, 2017). In Western Africa, some countries have introduced ISPs, which have increased the total use of fertilizer by 39 percent but with a reduction in the use of commercial fertilizer by 18 percent (Ricome, Barreiro-Hurle and Sadibou Fall, 2024). Access to subsidized fertilizer was associated with a reduction in the likelihood of using manure by 5 percent and an increase in farmers' total gross margin of 11 percent.

Currently, ISPs are heavily impacted by the Russian Federation–Ukraine conflict, which has made it unaffordable for some countries in sub-Saharan Africa to source fertilizers, including N fertilizers, due to restricted and disrupted global supply and skyrocketing prices of fertilizers (Nhlengethwa *et al.*, 2023). This situation has hindered the proper execution and implementation of ISPs and led to further reduced N application rates. This situation exacerbates the decline in agricultural productivity, particularly for farming systems burdened by low NUE or N deficiency (Abay *et al.*, 2023).

In response, some sub-Saharan African governments have increased fertilizer subsidies to support farmers. There is an increasing need to repurpose these subsidies towards the development and adoption of green innovations, such as the use of nitrification inhibitors and efficient practices that improve NUE and crop yield and incentivize the use of the right fertilizer type in the right places with minimum environmental impact (Gautam *et al.*, 2022).

### *Overview of agricultural policies and impacts on nitrogen use efficiency in Latin America and the Caribbean, the European Union and North America*

The average cropland NUE in European Union countries and North America varies between 66 and 69 percent (van Grinsven *et al.*, 2015), with these relatively high values being partly due to environmental and fertilizer regulations, as well as the economics of large-scale farming. Values of crop NUE have increased over the period 1961–2021 (FAO, 2023b), which echoes the value of the adoption of best practices of N fertilization and new genetic technologies (Drechsel *et al.*, 2015; Omara *et al.*, 2019). For instance, the European Union Nitrates Directive has led to slightly improved NUE by regulating manure and fertilizer use near waters (FAO, 2023b; Lassaletta *et al.*, 2014a), although the N pollution levels have not decreased much in certain countries (Oenema *et al.*, 2011). In North America, strategies such as the promotion of slow-released fertilizers, combined with increases in the cultivation of soybean and nitrogen-fixing plants, have contributed to increased NUE.

Latin America and the Caribbean (LAC) rely heavily on imported fertilizers, at the rate of approximately 85 percent. This dependency on imported fertilizers results in a high vulnerability to global price fluctuations, especially in the context of the Russian Federation–Ukraine conflict. Domestic fertilizer production has seen minimal growth, but policies to reduce fertilizer imports have started to emerge. For example, the Brazilian national fertilizer plan aims to reduce fertilizer imports by 45 percent by 2050 and promotes domestic fertilizer production and best practices in national production (The Government of Brazil, 2022). National initiatives supporting vulnerable farmers have been improved through the distribution of free bags of certified seeds and fertilizer in some countries, such as Honduras and Guatemala (Maredia, Reyes and DeYoung, 2014). Some countries in Latin America have incentivized the use of sustainable soil management practices to replace N fertilizer use. For instance, the Brazilian Plan for Low Carbon Emission in Agriculture provides resources and incentives for farmers who adopt methods that increase biological N fixation, no-till systems, methods that rehabilitate degraded pastureland, integrated crop–livestock–forestry systems, planted forests, and manure management systems (Pires *et al.*, 2015). These practices contribute to lowering N demand for crops for the following years of cultivation. Latin American and Caribbean countries still suffer from the lack of N-specific policies, and a common directive or framework in which nations can create their regulations (Zeri and Ometto, 2018).

### *Barriers to the adoption of agriculture technologies for enhanced nitrogen use efficiency*

Globally, many countries are promoting diverse agriculture technologies designed to improve yields and reduce the vulnerability of agricultural systems (Sitko, Scognamillo and Malevolti, 2021). Implementing best practices to ensure sustainable N management and increase crop NUE is challenging. Gu *et al*. (2023) highlighted the high heterogeneity of best agricultural practices at the local level. Those authors identified 11 key cost-effective measures that can significantly decrease N loss and increase crop yield and NUE. They vary from crop legume rotation and application of buffer zones to 4Rs nutrient stewardship and the introduction of new cultivars, optimal irrigation, and tillage (see Chapter 2). Furthermore, the use of organic fertilizer is seen as a "no regret" management strategy (Snapp *et al.*, 2023). The key often lies in customizing N management solutions and technology to the specifics of cropping systems and seasons in each region.

Several factors can constrain technology adoption, such as high input cost, poor targeting, low farmer education, poor market and credit access, small land size, lack of extension services, and land "tenure" (land tenure insecurity) insecurity (Suri and Udry, 2022). Addressing structural issues and barriers to adoption must be integral to national agrifood programmes and policies. Financial innovation mechanisms such as crop insurance can highly incentivize adoption because they protect farmers from losses and weather risks. Studies have indicated that increased farmer participation in the US crop insurance programmes is linked to lower N concentrations in waters (Lu *et al.*, 2023) and reduced N fertilizer application rates (Babcock and Hennessy, 1996). In the United States, there is ongoing reform of crop insurance policies since current ones can discourage the adoption of new practices (Annan and Schlenker, 2015) or restrict practices such as cover crop, crop intensification, or crop diversification, which impact the N cycle (Gelardi, Rath and Kruger, 2023). Information should be made available to farmers, who may have limited knowledge of N and do not know about new technologies or how to use them effectively. Extension, social networking, and technical assistance can help farmers gain technical knowledge on a variety of subjects, such as integrated soil fertility management (Khonje *et al.*, 2022), which can increase NUE and fertilizer savings.

Farmer's uptake of practices and technologies to improve NUE largely depends on whether the incentive is compulsory or voluntary (Schirmer, Dovers and Clayton, 2012). Regulatory policies, mandatory N management programmes, and financial incentives encourage compliance with sustainable practices and fertilizer use regulations more effectively than voluntary initiatives (Wood *et al.*, 2022) by imposing sanctions for non-compliance. An extensive survey identifying gaps and opportunities in N pollution around the world revealed that policies incentivizing N prevail over those regulating its use (Kanter *et al.*, 2020a). In the United States, both federal and state governments have largely eschewed direct regulations of agriculture, focusing instead on voluntary and incentives-based tools (Reimer, Denny and Stuart, 2018). The voluntary incentive programmes do not encourage compliance and have a high degree of uncertainty because the practice's adoption depends solely on farmers' decisions. Farmers tend to prefer voluntary measures over regulations, especially if they provide economic incentives as well (Piñeiro *et al.*, 2020). This approach brings a significant level of uncertainty regarding the attainment of the programme's environmental objectives. In the European Union, to meet the targets of the Farm to Fork strategy – aiming to decrease nutrient losses to the environment by a minimum of 50 percent and reduce fertilizer usage by at least 20 percent by 2030 – Wassen *et al*. (2022) proposed the implementation of an integrated nutrient directive. This directive would regulate the agricultural application of N and P, ensuring a more certain path toward achieving its goals. The political sensitivity surrounding the agricultural sector, the existence of non-point and point N pollution sources, and N's dual role as a pollutant and nutrient make regulating the agricultural sector challenging (Kanter *et al.*, 2020a). Yang *et al.* (2021) analysed the effects of economic incentives and regulatory restrictions on the reduction of synthetic fertilizer in China. They found that combining both could effectively encourage sustainable production behaviour among farmers.

Countries can offer incentives and subsidies to farmers to adopt best management practices. Subsidies can integrate cross-compliance incentives to help compensate for the income loss or additional costs of adopting management practices while farmers meet certain environmental standards and clear monitoring practices. In the European Union, cross-compliance is fundamental to the Common Agricultural Policy. It requires farmers to comply with various policies related to the environment, food safety, animal and plant health, and animal welfare, among others. This compliance is essential for farmers to qualify for direct payments to support agricultural income (Kanter *et al.*, 2020b).

### 6.1.2 Partnerships and initiatives to tackle the nitrogen challenge

Nitrogen figures prominently in policy initiatives, including the sustainable N management resolutions of the United Nations Environmental Assembly (UNEP, 2019a), the 2019 Colombo Declaration, the Global Partnership for Nutrient Management, the International Nitrogen Initiative, and the Global Soil Partnership Initiatives. These initiatives emphasize the need for a coordinated global effort to manage N effectively, reflecting the growing awareness of the impact of N on ecosystems and human well-being. The S*tatus of the World's Soil Resources* report (FAO and ITPS, 2015), the *Voluntary Guidelines for Sustainable Soil Management* (FAO, 2017), and the *International Code of Conduct for the Sustainable Use and Management of Fertilizers* (Fertilizer Code) (FAO, 2019) identified nutrient imbalance, especially N, as one of the top ten threats to soil and human health. These reports provide recommendations to tackle the causes and consequences of nutrient-overloaded or undernourished soils.

## 6.2 NITROGEN IN LIVESTOCK DEVELOPMENT POLICIES

### 6.2.1 Nitrogen and manure in environmental policies

Livestock supply chains are responsible for a substantial share, estimated at approximately one-third of human-induced reactive N emissions (Uwizeye *et al.*, 2020). In addition to emissions from feed production, livestock contribute directly to emissions through N losses from manure management systems and applied and deposited manure. Manure is a valuable resource and provides a source of minerals and organic matter to soil and crops.

Globally, there are regional disparities in manure policies. In the European Union, manure management regulations to reduce N pollution have been designed, established and enforced to comply with Nitrates Directive, Water Framework Directive, Air Quality Directive, and Directives on livestock production, manure, and biowaste management and environmental protection (Sommer *et al.*, 2013). Excessive use of manure has been regulated by application standards based on N and P content. As such, regulations prohibit the discharge of manure into surface waters and limit the N application from manure to 170 kg N/ha/yr when part of a nationally declared "nitrate vulnerable zone" (Landbrugsstyrelsen, 2019). In some countries of Europe, such as Denmark, farmers must submit comprehensive fertilization plans for each field and adhere to a minimum standard for the efficiency of manure N utilization (Sommer and Knudsen, 2021). Similarly, Danish farmers are required to utilize techniques for low-emission storage, handling and application of manure, reducing $NH_3$ emissions. This is not required in all European Union Member States. Regulations on manure storage capacity apply for $NO_3^-$-vulnerable zones as part of the Nitrates Directive, such as the construction of livestock production units and of stores for manure (minimum nine months' storage capacity) with impermeable floors to limit liquid leaching and restrictions on time and techniques for manure application, are implemented across European Union countries (Oenema, 2004; Sommer and

Knudsen, 2021). The extent and impacts of policies and regulations vary significantly among Member States due to the great diversity of environmental conditions (Oenema, 2004), such as geomorphology and farmers' adaptation capacity to legislation (Méité, Artner-Nehls and Uthes, 2024). Farmers are encouraged to use low-protein animal feeds and reduce $NH_3$ emissions by low-emission animal housing and manure storage systems and by low-emission manure application (through injection, trailing hose, or rapid incorporation into the soil) with all European Union Member States required to establish a National Ammonia Code (of voluntary measures) under the terms of Annex IX of the United Nations Economic Commission for Europe (UNECE) Gothenburg Protocol (UNECE, 1999). Progress in establishing such national codes across the UNECE region has been slow – see ECE/EB.AIR/WG.5/2010/13, paragraph 33 (UNECE, 2010) – although the number is increasing.

In sub-Saharan African countries, manure management policies lack coherence due to their fragmented design and implementation across various ministries and are seldom enforced because of weak coordination between ministries and their enforcing bodies (Teenstra *et al.*, 2014). These policies often view manure as a waste product, prioritizing its management for human health and pollution concerns rather than recognizing its fertilizer value (Holden and Lunduka, 2012; Ndambi *et al.*, 2019). As a result, farmers may feel discouraged from using manure as a fertilizer, which is labour-intensive to collect, prepare and apply. Increased government subsidies for synthetic fertilizers in sub-Saharan African may discourage the use of manure (Holden and Lunduka, 2012). For instance, Ketema and Bauer (2011) found that farmers in Ethiopia who can afford synthetic fertilizers often use less manure. Conversely, incentives could be used to promote manure usage, such as the case for certain conservation programmes within the US Farm Bill that incentivize farmers to adopt conservation practices, including manure management systems (Tomich *et al.*, 2016).

Mineral N fertilizer can be recovered from manure through processing (Huygens *et al.*, 2020). These recovered N fertilizer products showed similar performance to synthetic fertilizers in terms of NUE, crop yields, and $NO_3^-$ leaching (Reuland *et al.*, 2021; Saju *et al.*, 2023). They increase microbial diversity and stability, indicative of healthy, productive soils (Saju *et al.*, 2023). Good management practices must be adopted to reduce potential $NH_3$ and $N_2O$ emissions and $NO_3^-$ leaching from recovered N fertilizer (Huygens *et al.*, 2020).

### 6.2.2 Policies to support the recycling of food losses and waste as animal feed

Repurposing FLW into animal feed offers multiple benefits, such as reducing the amount of FLW in landfills, minimizing GHG emissions and nutrient footprint of food production, contributing to a circular bioeconomy, and other co-benefits. The concept of FLW is self-contradictory, leading to policy incoherence and restrictions. While "food" is positive, desirable and healthy, "waste and loss" are negative, undesirable and unhealthy (Marouli, 2024). Because of this association with "waste and loss", policymakers often perceive FLW as a problem and a risk that requires management, rather than recognizing it as a valuable resource that needs to be recycled into high-value purposes such as animal feed. As such, terms like "swill" and "garbage feeding" carry negative connotations and fail to acknowledge the nutritional and economic value of using FLW in feeding animals (Dou, Toth and Westendorf, 2018; Shurson, 2020). Moreover, infectious diseases such as African swine fever and foot-and-mouth disease are potentially transmitted, which prohibit their wide adoption as feed (Uwizeye *et al.*, 2019). Breaking this perception is the first barrier to overcome to encourage the use of FLW as animal feed. Re-evaluating policies that restrict or prohibit FLW repurposing, such as those in North America and Europe, is imperative to alleviate these barriers.

Only a small portion of FLW produced in North America and Europe is used in animal feed, amounting to approximately 5 percent and 10 percent, respectively (Boumans *et al.*, 2022; McBride *et al.*, 2021). In contrast, a significant share of FLW is used as feed in Japan (36 percent) and Republic of Korea (43 percent) (zu Ermgassen *et al.*, 2016; Takata *et al.*, 2012; Uwizeye *et al.*, 2019). These countries have developed innovative policies and regulations to collect, prepare, heat-treat, and improve FLW traceability (zu Ermgassen *et al.*, 2016; Liu *et al.*, 2016; Sugiura *et al.*, 2009; Takata *et al.*, 2012; Uwizeye *et al.*, 2019). Along with restrictive policies and legal framework, the lack of economic incentives and infrastructure for collecting, transporting and processing different FLW in animal feed creates barriers. For instance, Marouli (2024) conducted a case studies analysis and found that the way cities are organized spatially and temporally poses challenges to reducing and recycling FLW. The overall lack of public awareness about the scale and environmental and economic impacts of the FLW worsens the situation (Vanham *et al.*, 2019).

While agriculture policies are centred on farm-level management of N, many actors in the agrifood chain contribute to generating N pollution and benefit from its reduction, including fertilizer manufacturers and wastewater treatment industries. All actors in the agrifood chain should be jointly responsible for supporting a decrease in N loss and sharing N abatement costs and benefits (Sutton *et al.*, 2022). As such, there is an urgent need for policy interventions that could improve N management for major agrifood chain actors while influencing farm-level decisions (Kanter *et al.*, 2020a). Such policies include encouraging diets with low N footprints and household composting. Also, beyond the farm, policies and strategies to reduce food waste along the agrifood chain are needed to offset N pollution. This is because a large fraction of N fertilizers is wasted in the food that is not consumed (Houlton *et al.*, 2019). For instance, only 45 percent of the N flows in the post-farm gate food system end up as ingested N (Corrado *et al.*, 2020). Increasing N recycling and raising public awareness to curb excessive purchases and composting will help decrease N issues associated with food waste.

## 6.3 NITROGEN IN POLICIES RELATED TO ORGANIC RESIDUES AND WASTE

Nitrogen flows in organic wastes and other residues originate from domestic and industrial sources. They comprise two primary categories: solid waste (such as discarded food products and packaging), and wastewater and sewage sludge. Notably, sewage, wastewater and food waste have a high N content, with food waste containing around 16 percent (Reis *et al.*, 2016). In contrast to agriculture, most policies addressing N in the organic waste sector are regulatory (Kanter *et al.*, 2020b).

Management of wastewater and sewage is critical to minimize N impacts on the environment and promote N circularity. Sewage and wastewater account for 3 percent of global $N_2O$ emissions, mainly originating from the direct discharge of wastewater effluent and the release of $N_2O$ during the biological removal of N by bioreactors (Davidson and Kanter, 2014). In the European Union, the establishment of wastewater treatment facilities is mandated by the Urban Wastewater Treatment Directive, known as Community Directive 91/271/EEC. According to this directive, "sludge arising from wastewater treatment shall be re-used whenever appropriate". Sludge is often used as organic fertilizer based on its N content, absence of heavy metals, and compliance with the Directive on Sewage Sludge. This directive aims to ensure surface and groundwater are not contaminated with N originating from sludge, regulate the documentation of sludge usage, and provide recommendations for N and nutrient needs of plants, quality of soil, and sludge stabilization. Some countries, such as Belgium, Germany and the Kingdom of the Netherlands, prohibit land application of sewage sludge by legislation and prefer thermal disposal methods (Bauer *et al.*, 2020).

In Latin American and Caribbean countries, resource recovery (including N) is often restricted by regulations. For example, in Peru, sewage sludge is classified as hazardous solid waste and must be disposed of in a secure cell within a sanitary landfill (Martin-Hurtado and Nolasco, 2017). Latin American and Caribbean countries typically lack technical capabilities in the sludge stabilization process, and sludge rarely goes through adequate disposal (Laura *et al.*, 2020). There is a tendency to favour costly technologies, such as activated sludge, for stabilizing sludge (Martin-Hurtado and Nolasco, 2017). Conversely, some countries like Brazil and Argentina have a supportive regulatory environment, such as the Brazilian Resolution (CONAMA, 2020), which establishes thresholds for biosolid (i.e. treated sludge) quality to allow application in soils (Garbellini *et al.*, 2023). The Argentinian regulations foster the sustainable use of biosolids from wastewater plants (Argentina Ambiental, 2018) to reduce the impacts of potentially toxic elements (Lavado, Rodríguez and Taboada, 2005).

Food waste is a critical source of N that requires management. Globally, 1.05 billion tonnes/yr of food waste are generated (UNEP, 2024). Interestingly, higher per capita rates of food waste are observed in Europe and Northern America (95–115 kg/yr), while lower rates are seen in sub-Saharan Africa and South and Southeast Asia (6–11 kg/yr) (FAO, 2011). Food loss and waste constitutes the largest share of municipal solid waste in Latin American and Caribbean countries (50 percent) (Ulloa-Murillo *et al.*, 2022). Food loss and waste occurring at the consumption level contributes to the annual loss of 2.7 Tg N globally and delivers about 6.3 Tg N to the environment (Grizzetti *et al.*, 2013). Reducing N originating from FLW and improving NUE along the food chain is critical. Current policies regarding FLW vary widely among countries. For instance, China is focused on reducing production loss and enhancing supply chain efficiencies (Joshi and Visvanathan, 2019). Republic of Korea has implemented pay-as-you-throw systems for household food waste and landfill and incineration bans (Richa and Ryen, 2018). In the United States, many laws have been enacted to prevent FLW, including the Food Loss and Waste Reduction Goal (EPA, 2024), which aims to reduce FLW generation and landfilling by 50 percent by 2030. Similarly, the European Union strives to halve N loss, which may need a decrease in FLW and dietary changes (Leip *et al.*, 2022). In Latin American and Caribbean countries, FLW reduction is in the action plan for Food and Nutrition Security and the Eradication of Hunger 2025 of the Community of Latin American and Caribbean States, which is the main political forum of the region. The region has seen the development of policies regarding FLW reduction in many countries, such as Argentina (e.g. National Programme for Food Loss and Reduction) and Brazil (e.g. Zero Hunger programmes) as well as the development of food banks (FAO, 2015). FAO has developed a voluntary international code of conduct for FLW reduction, which serves as internationally recognized guiding principles to reduce FLW effectively. Countries can adapt these principles for the development of local and national strategies, policies and programmes that focus on FLW reduction (FAO, 2022c).

## 6.4 SUSTAINABLE NITROGEN MANAGEMENT AND INTERNATIONAL POLICY AGENDA

Sustainable N management is interlinked with the 2030 Agenda for Sustainable Development. Increasing NUE across all parts of the agrifood chain and reducing N losses contribute directly to the achievement of SDG 2 on ending hunger, SDG 3 on improving health and well-being, SDG 6 on clean water and sanitation, SDG 12 on sustainable consumption and production, SDG 13 on climate action, SDG 14 on life underwater and SDG 15 on life on land. Regarding SDG 2, the improvement of access to N inputs can boost food production in LMICs, thus reducing the number of people facing hunger and food insecurity, which has been exacerbated by conflict, climate change, and increasing inequality (UN, 2023). Urgent action and policy are needed to accelerate the sustainable transformation of agrifood systems to achieve food security and nutrition, leaving no one behind. Both $NH_3$ and $NO_x$ emissions react with other chemicals to form particulate matter, which increases risks for respiratory and heart diseases. Nitrogen oxides emissions contribute to tropospheric $O_3$ formation, which reduces crop yields, pointing to the imperative to reduce these emissions. Both chemicals are harmful to human health, causing respiratory diseases and death (Independent Group of Scientists appointed by the Secretary-General, 2019; Sutton *et al.*, 2021). Reducing N runoff and leaching to water bodies and recovery of N from wastewater is essential to achieve SDG 6, particularly the Indicators 6.3.1 on a proportion of wastewater safely treated and 6.3.2. on a proportion of bodies of water with good ambient water quality (Sutton *et al.*, 2021). Better management of N is important to achieve SDG 12.2 on achieving sustainable management and efficient use of natural resources, SDG 12.4 on achieving the environmentally sound management of chemicals and all wastes, and SDG1 2.5 on substantially reducing waste generation through prevention, reduction, recycling and reuse. Moreover, it can contribute to SDG 14 and SDG 15 by reducing freshwater and marine eutrophication and terrestrial and freshwater acidification (Independent Group of Scientists appointed by the Secretary-General, 2019; Vanham *et al.*, 2019). Achieving SDG 13 in agrifood systems can build on the reduction of $N_2O$ emissions through the integration of sustainable N management strategies in national climate change measures and policies (Independent Group of Scientists appointed by the Secretary-General, 2019; Sutton *et al.*, 2021). Nevertheless, reducing $N_2O$ emissions can help to achieve the Paris Agreement objectives to limit the increase of the average global temperature to well below 2 °C and preferably to below 1.5 °C (IPCC, 2023).

Furthermore, the United Nations Environmental Assembly has adopted two resolutions calling for increased action to address sustainable N management in 2019 (UNEP, 2019b) and 2022 (UNEP, 2022). These resolutions aim to encourage a significant reduction of global N waste by 2030 and emphasize the ambition outlined in the Colombo Declaration (UNEP, 2019b). Target 7 of the Kunming-Montreal Global Biodiversity Framework calls for "reducing pollution, including reducing excess nutrients lost to the environment by at least half including through more efficient nutrient cycling and use", and Target 16 calls for "halving global food waste by 2030 and significantly reduce over-consumption", both linking to sustainable N management and reducing N losses to the environment (CBD, 2022).

For these efforts to succeed, policies need to reconcile the dual role of N as an important nutrient necessary for economic growth, human advancement and food security, and a pollutant that causes serious ecosystem damage. Recognizing this fundamental challenge, FAO's Global Soil Partnership positions soils at the centre of the SDGs, ensuring the Earth's ecosystems are healthy with healthy soils. The overuse and misuse of N fertilizers coupled with unsustainable soil management, leads to a leaky N cycle and threatens soil health (see Chapter 2). FAO has established the International Code of Conduct for the Sustainable Use and Management of Fertilizers, known as the Fertilizer Code (FAO, 2019). It is a useful instrument that provides recommendations related to the use of recycled nutrient sources to increase food safety and the safe use of fertilizers. It provides a locally adaptable framework and a voluntary set of practices relevant to different stakeholders involved with fertilizer production, distribution and use.

Furthermore, in the global roadmap to achieving SDG 2 without breaching the 1.5 °C threshold, FAO has presented the circular bioeconomy as an opportunity to achieve the SDG 12 targets of reducing food loss and waste by 50 percent and 100 percent by 2030 (FAO, 2023d).

## 6.5 KEY ACTIONS AND POLICY OPTIONS TO PROMOTE SUSTAINABLE NITROGEN MANAGEMENT

### *Increase best practices in mineral fertilizer production and use and enhance soil health in cropping systems*

- The fertilizer industry should urgently take action to cut GHG emissions during the production of mineral N fertilizer and promote the reduction of wasteful losses during storage, transport and application to the land. In LMICs, measures to enhance access to high-quality mineral fertilizers while mitigating the environmental impacts associated with their use are needed.
- Countries should promote the widespread use of biological N fixation in locally suited crop rotations and increase leguminous crops, which represent additional integrative approaches to increasing NUE while reducing regional pollution. Where possible, they should implement agroecological practices such as strip cropping, cover cropping, and conservation agriculture to increase soil nutrient status and health, restore degraded land, reduce erosion, and minimize N losses. Moreover, the elaboration of national fertilizer use recommendations based on the 4Rs approach (right rate, right source, right time, right place) in line with the International Code of Conduct

for the Sustainable Use and Management of Fertilizers (FAO, 2019) and the Voluntary Guidelines for Sustainable Soil Management (FAO, 2017).

- Countries should encourage, as appropriate, the use of biosolids, including sewage across scales, and demand-side approaches that address consumption patterns to promote highly efficient fibre, food and bioenergy production.

***Improve nitrogen management in the livestock sector***

- Countries should develop guidelines to help livestock farmers adopt the best manure management techniques, with a focus on reducing wasteful N losses to the environment and maximizing their effective use in productive agriculture. The improvement of manure management systems, including liquid–solid segregation and separation, covered manure storage, and low-emission manure application to land to reduce the volatilization of $NH_3$ and enhance the retention of nutrients for crop production should be promoted.
- Livestock producers and farmers should adopt measures to improve feed formulation and feed use efficiency by optimizing protein intake, which reduces N excretion for different livestock species and further N losses to the environment.
- Countries should implement policies that enhance spatial integration of crop and livestock production through agroecological approaches. These approaches implement circular bioeconomy principles at the landscape level, maximizing the opportunity for effective nutrient reuse.
- Countries should bolster more efforts to reduce FLW at all stages of the agrifood systems and promote the recycling and treatment of food unsuitable for human consumption as livestock feed.

***Promote public and private investment***

- National governments, the private sector, international financial institutions, and local agricultural banks should mainstream sustainable N management into development projects and programmes in agrifood systems and promote investment in high-efficiency, low-emission mineral fertilizers and the recycling of organic residues to enhance system efficiency, reduce resource waste, and reduce environmental pollution.
- Agrifood system stakeholders should promote, as appropriate, the investment in agroecology and sustainable crop–livestock integrated development projects to enhance sustainable N management. Moreover, the public, private sector, producers' organizations, non-government organizations, and academia should engage to promote sustainable N management across the crop and livestock value chains.

***Capacity building at scale***

- Countries and international development partners should support national capacity building on sustainable N management of different agrifood system stakeholders, including the public, private sector, civil society organizations, and farmers and producer organizations, and strengthen national extension services, research and knowledge transfer as well as promote sustainable N management practices through farmer field schools. Essential is the promotion of wider learning between different visions for sustainable N management, improving mutual understanding of the merits and risks of different strategies, including farming systems focused on fertilizer inputs, agroecology, and regenerative farming approaches.

***Policy options***

- Agrifood system policies should encourage the use of manure as a source of organic N fertilizer to enhance sustainability. They should also implement various efforts to improve spatial planning, redistribute livestock, and reduce livestock numbers in areas with high geographical concentration to levels of balanced crop and livestock integration. Demand-side policies should focus on addressing consumption patterns and promoting low N emission diets, while taking into account other environmental impacts.
- Countries should promote the integration of sustainable N management in nationally appropriate mitigation actions and nationally determined contributions, including targets to reduce $N_2O$ from agrifood systems to keep the Paris Agreement goal of 1.5 °C in sight. Moreover, countries should set national commitments to reduce N pollution, including $NH_3$ and $NO_x$ emissions to air and $NO_3^-$ losses to water bodies, in line with Target 7 of the Kunming-Montreal Biodiversity Framework and SDGs 6, 12, 13, 14, 15 and 17.
- Countries and agrifood system stakeholders should develop market-based incentives, such as premium prices and other instruments, to reward farmers who comply with sustainable N management and environmental regulations. In areas with high livestock concentrations, the development of country-specific manure management policies to promote a circular bioeconomy and reduce environmental pollution, such as the discharge of manure in water bodies or landfills, should be prioritized.
- Countries should promote partnership and international cooperation to address N access and pollution globally through different intergovernmental fora.

# Conclusion

The report covers areas related to NUE in cropping and livestock systems, the impact of N losses on ecosystems, circular bioeconomy, and policy instruments to promote sustainable N management in agrifood systems. It underscores the need for joint responsibility among agrifood chain actors to decrease N loss and share abatement costs. It advocates for policies that encourage low N emission diets, sustainable use of fertilizers, circular bioeconomy, better manure management systems, and strategies to reduce food waste and loss beyond the farm level. This is crucial as a large portion of N fertilizers is wasted in unconsumed food, highlighting the importance of increasing N recycling and public awareness to address N-related issues associated with food waste. Moreover, there is a growing trend to recommend the replacement of synthetic fertilizers with organic fertilizers, but this must be done with adequate quality control and proper rate and application methods. Soil health is important, so no fertilizer application will be effective if soils are compacted, eroded, dried, acidified, with low soil biodiversity, or with a low SOM content.

Sustainable N management is crucial for achieving the Sustainable Development Goals by 2030, particularly those related to hunger, health, clean water, sustainable production and consumption, climate action, and preserving life on land and underwater. Improving NUE across the agrifood chain and reducing N loss can help increase food production in LMICs by allowing more N resources to achieve their intended purpose, improve health by reducing harmful emissions, and protect water bodies from pollution. Key actions identified to promote sustainable N management include increasing sustainable fertilizer production and improving soil health in LMICs, improving manure management in the livestock sector, and promoting public and private investment in sustainable agrifood systems. Capacity building and policy development are essential, with a focus on integrating sustainable N management into national climate policies, developing market-based incentives and fostering international cooperation to address N access and pollution. These efforts align with global targets such as the Kunming-Montreal Biodiversity Framework, the United Nations Agenda 2030 for Sustainable Development, and the Paris Agreement, aiming to reduce pollution and food waste and to manage resources more efficiently for a sustainable future.

# References

**Abalos, D., van Groenigen, J.W. & de Deyn, G.B.** 2018. What plant functional traits can reduce nitrous oxide emissions from intensively managed grasslands? *Global Change Biology*, 24(1): 248–258. https://doi.org/10.1111/gcb.13827

**Abalos, D., Jeffery, S., Sanz-Cobena, A., Guardia, G. & Vallejo, A.** 2014. Meta-analysis of the effect of urease and nitrification inhibitors on crop productivity and nitrogen use efficiency. *Agriculture, Ecosystems & Environment*, 189: 136–144. https://doi.org/10.1016/j.agee.2014.03.036

**Abascal, E., Gómez-Coma, L., Ortiz, I. & Ortiz, A.** 2022. Global diagnosis of nitrate pollution in groundwater and review of removal technologies. *Science of The Total Environment*, 810: 152233. https://doi.org/10.1016/j.scitotenv.2021.152233

**Abay, K., Chamberlin, J., Chivenge, P., Hebebrand, C. & Spielman, D.J.** 2023. Fertilizer policies amid global supply and price shocks. [Cited 15 March 2024]. https://www.ifpri.org/blog/fertilizer-policies-amid-global-supply-and-price-shocks/

**Abdullahi, A.A., Aliyu, I.A. & Gabasawa, A.I.** 2020. Symbiotic $N_2$ fixation as an alternative source of nitrogen – A review. *Ife Journal of Agriculture*, 32(2): 36–45.

**Aber, J., McDowell, W., Nadelhoffer, K., Magill, A., Berntson, G., Kamakea, M., McNulty, S. *et al.*** 1998. Nitrogen saturation in temperate forest ecosystems: Hypotheses revisited. *BioScience*, 48(11): 921–934. https://doi.org/10.2307/1313296

**Abu-alnaeem, M.F., Yusoff, I., Ng, T.F., Alias, Y. & Raksmey, M.** 2018. Assessment of groundwater salinity and quality in Gaza coastal aquifer, Gaza Strip, Palestine: An integrated statistical, geostatistical and hydrogeochemical approaches study. *Science of the Total Environment*, 615: 972989. https://doi.org/10.1016/j.scitotenv.2017.09.320

**African Union & AUDA-NEPAD (African Union Development Agency – New Partnership for Africa's Development)**. 2024. *Comprehensive African Agricultural Development Programme (CAADP)*. [Cited 25 March 2024]. https://caadp.org

**Agyekum, T.P., Antwi-Agyei, P. & Dougill, A.J.** 2022. The contribution of weather forecast information to agriculture, water and energy sectors in East and West Africa: A systematic review. *Frontiers in Environmental Science*, 10: 935696. https://doi.org/10.3389/fenvs.2022.935696

**Ahmadzai, H., Tutundjian, S., Dale, D., Brathwaite, R., Lidderr, P., Selvaraju, R., Malhotra, A., Boerger, V. & Elouafi, I.** 2022. *Marginal lands: Potential for agricultural development, food security and poverty reduction*. Rome, Food and Agriculture Organization of the United Nations. https://openknowledge.fao.org/handle/20.500.14283/cc2838en

**Alecrim, F.B., Alves, B.J.R., Rezende, C. de P., Boddey, R.M., Nobrega, G.N., Cesário, F.V., Sobral, B.S. *et al.*** 2023. The influence of tropical pasture improvement on animal performance, nitrogen cycling and greenhouse gas emissions in the Brazilian Atlantic Forest. *Australian Journal of Crop Science*, 17(4): 392–399. http://dx.doi.org/10.21475/ajcs.23.17.04.p3824

**Altieri, A.H. & Gedan, K.B.** 2015. Climate change and dead zones. *Global Change Biology*, 21(4): 1395–1406. https://doi.org/10.1111/gcb.12754

**Amann, M., Bertok, I., Borken-Kleefeld, J., Cofala, J., Heyes, C., Höglund-Isaksson, L., Klimont, Z. *et al.*** 2011. Cost-effective control of air quality and greenhouse gases in Europe: Modelling and policy applications. *Environmental Modelling & Software*, 26(12): 1489–1501. https://doi.org/10.1016/j.envsoft.2011.07.012

**Amon, B., Kryvoruchko, V., Amon, T. & Zechmeister-Boltenstern, S.** 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. *Agriculture, ecosystems & environment*, 112(2–3): 153–162. https://doi.org/10.1016/j.agee.2005.08.030

**Anglade, J., Billen, G., Garnier, J., Makridis, T., Puech, T. & Tittel, C.** 2015. Nitrogen soil surface balance of organic vs conventional cash crop farming in the Seine watershed. *Agricultural Systems*, 139: 82–92. https://doi.org/10.1016/j.agsy.2015.06.006

**Anglade, J., Billen, G. & Garnier, J.** 2017. Reconquérir la qualité de l'eau en régions de grandes cultures : agriculture biologique et reconnexion avec l'élevage. *Fourrages*, 231: 257-268. https://afpf-asso.fr/article/reconquerir-la-qualite-de-l-eau-en-regions-de-grandes-cultures-agriculture-biologique-et-reconnexion-avec-l-elevage

**Annan, F. & Schlenker, W.** 2015. Federal crop insurance and the disincentive to adapt to extreme heat. *American Economic Review*, 105(5): 262–266. http://dx.doi.org/10.1257/aer.p20151031

**Aramburu-Merlos, F., van Loon, M.P., van Ittersum, M.K. & Grassini, P.** 2024. High-resolution global maps of yield potential with local relevance for targeted crop production improvement. *Nature Food*: 1-6. https://doi.org/10.1038/s43016-024-01029-3

**Arnés, M. & Santiváñez, T.** 2021. *Hand in hand with nature – Nature-based solutions for transformative agriculture*. Budapest, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cb4934en

**Babcock, B.A. & Hennessy, D.A.** 1996. Input demand under yield and revenue insurance. *American Journal of Agricultural Economics*, 78(2): 416–427. https://doi.org/10.2307/1243713

**Bacenetti, J., Paleari, L., Tartarini, S., Vesely, F.M., Foi, M., Movedi, E., Ravasi, R.A. *et al.*** 2020. May smart technologies reduce the environmental impact of nitrogen fertilization? A case study for paddy rice. *Science of the Total Environment*, 715: 136956. https://doi.org/10.1016/j.scitotenv.2020.136956

**Bai, Y., Wu, J., Clark, C.M., Naeem, S., Pan, Q., Huang, J., Zhang, L. & Han, X.** 2010. Tradeoffs and thresholds in the effects of nitrogen addition on biodiversity and ecosystem functioning: Evidence from inner Mongolia grasslands. *Global Change Biology*, 16(1): 358372. https://doi.org/10.1111/j.1365-2486.2009.01950.x

**Bai, Z., Ma, L., Jin, S., Ma, W., Velthof, G.L., Oenema, O., Liu, L., Chadwick, D. & Zhang, F.** 2016. Nitrogen, phosphorus and potassium flows through the manure management chain in China. *Environmental Science & Technology*, 50(24): 13409–13418. http://dx.doi.org/10.1021/acs.est.6b03348

**Bai, Z., Fan, X., Jin, X., Zhao, Z., Wu, Y., Oenema, O., Velthof, G., Hu, C. & Ma, L.** 2022. Relocate 10 billion livestock to reduce harmful nitrogen pollution exposure for 90 percent of China's population. *Nature Food*, 3(2): 152–160. http://dx.doi.org/10.1038/s43016-021-00453-z

**Baker, J., Bernard, D., Christensen, S., Sale, M., Freda, J., Heltcher, K., Rowe, L. *et al.*** 1990. Biological effects of changes in surface water acid-base chemistry. ORNL/M-1086, DE90008392. https://doi.org/10.2172/7255574

**Baker, E., Kerr, R.B., Deryng, D., Farrell, A., Gurney-Smith, H. & Thornton, P.** 2023. Mixed farming systems: Potentials and barriers for climate change adaptation in food systems. *Current Opinion in Environmental Sustainability*, 62: 101270. http://dx.doi.org/10.1016/j.cosust.2023.101270

**Banzhaf, H.S., Burtraw, D., Evans, D. & Krupnick, A.** 2006. Valuation of natural resource improvements in the Adirondacks. *Land Economics*, 82(3): 445–464. https://doi.org/10.3368/le.82.3.445

**Barbieri, P., Pellerin, S. & Nesme, T.** 2017. Comparing crop rotations between organic and conventional farming. *Scientific Reports*, 7(1): 13761. https://doi.org/10.1038/s41598-017-14271-6

**Bauer, T., Ekman Burgman, L., Andreas, L. & Lagerkvist, A.** 2020. Effects of the different implementation of legislation relating to sewage sludge disposal in the EU. *Detritus*. https://doi.org/10.31025/2611-4135/2020.13944

**Beal, T., Gardner, C.D., Herrero, M., Iannotti, L.L., Merbold, L., Nordhagen, S. & Mottet, A.** 2023. Friend or foe? The role of animal-source foods in healthy and environmentally sustainable diets. *The Journal of Nutrition*, 153(2): 409–425. https://doi.org/10.1016/j.tjnut.2022.10.016

**Beed, F., Benedetti, A., Cardinali, G., Chakraborty, S., Dubois, T., Halewood, M. & Garrett, K.A.** 2011. Climate change and micro-organism genetic resources for food and agriculture: state of knowledge, risks and opportunities. Background Study Paper No. 57. [Cited 3 March 2024] https://www.fao.org/4/mb392e/mb392e.pdf

**Beig, G., Maji, S., Panicker, A. & Sahu, S.** 2017. Reactive nitrogen and air quality in India. *The Indian Nitrogen Assessment*: 403–426. https://doi.org/10.1016/B978-0-12-811836-8.00025-2

**Beillouin, D., Pelzer, E., Baranger, E., Carrouée, B., Cernay, C., De Chezelles, E., Schneider, A. & Jeuffroy, M.-H.** 2021. Diversifying cropping sequence reduces nitrogen leaching risks. *Field Crops Research*, 272(8): 108268. http://dx.doi.org/10.1016/j.fcr.2021.108268

**Bell, L.W. & Moore, A.D.** 2012. Integrated crop-livestock systems in Australian agriculture: Trends, drivers and implications. *Agricultural Systems*, 111: 1–12. https://doi.org/10.1016/j.agsy.2012.04.003

**Beman, J.M., Arrigo, K.R. & Matson, P.A.** 2005. Agricultural runoff fuels large phytoplankton blooms in vulnerable areas of the ocean. *Nature*, 434(7030): 211–214. https://doi.org/10.1038/nature03370

**Beman, J.M., Chow, C.-E., King, A.L., Feng, Y., Fuhrman, J.A., Andersson, A., Bates, N.R., Popp, B.N. & Hutchins, D.A.** 2011. Global declines in oceanic nitrification rates as a consequence of ocean acidification. *Proceedings of the National Academy of Sciences*, 108(1): 208–213. https://doi.org/10.1073/pnas.1011053108

**Benoit, M., Garnier, J., Billen, G., Tournebize, J., Gréhan, E. & Mary, B.** 2015. Nitrous oxide emissions and nitrate leaching in an organic and a conventional cropping system (Seine basin, France). *Agriculture, Ecosystems & Environment*, 213: 131–141. https://doi.org/10.1016/j.agee.2015.07.030

**Bhattacharyya, P., Bisen, J., Bhaduri, D., Priyadarsini, S., Munda, S., Chakraborti, M., Adak, T. *et al.*** 2021. Turn the wheel from waste to wealth: Economic and environmental gain of sustainable rice straw management practices over field burning in reference to India. *Science of the Total Environment*, 775: 145896. https://doi.org/10.1016/j.scitotenv.2021.145896

**Bijay-Singh**. 2022. Nitrogen use efficiency in crop production in India: Trends, issues and challenges. *Agricultural Research* 12: 32–44. https://doi.org/10.1007/s40003-022-00626-7

**Billen, G., Lassaletta, L. & Garnier, J.** 2015. A vast range of opportunities for feeding the world in 2050: Trade-off between diet, N contamination and international trade. *Environmental Research Letters*, 10(2): 025001. https://doi.org/10.1088/1748-9326/10/2/025001

**Billen, G., Beusen, A., Bouwman, L. & Garnier, J.** 2010. Anthropogenic nitrogen autotrophy and heterotrophy of the world's watersheds: past, present, and future trends. *Global Biogeochemical Cycles*, 24: GB0A11. http://dx.doi.org/10.1029/2009GB003702

**Billen, G., Le Noë, J. & Garnier, J.** 2018. Two contrasted future scenarios for the French agro-food system. *Science of the Total Environment*, 637: 695–705. https://doi.org/10.1016/j.scitotenv.2018.05.043

**Billen, G., Aguilera, E., Einarsson, R., Garnier, J., Gingrich, S., Grizzetti, B., Lassaletta, L., Le Noë, J. & Sanz-Cobena, A.** 2021. Reshaping the European agro-food system and closing its nitrogen cycle: The potential of combining dietary change, agroecology and circularity. *One Earth*, 4(6): 839–850. https://doi.org/10.1016/j.oneear.2021.05.008https://doi.org/10.1016/j.oneear.2021.05.008

**Billen, G., Aguilera, E., Einarsson, R., Garnier, J., Gingrich, S., Grizzetti, B., Lassaletta, L., Le Noë, J. & Sanz-Cobena, A.** 2024. Beyond the farm to fork strategy: Methodology for designing a European agro-ecological future. *Science of the Total Environment*, 908: 168160. https://doi.org/10.1016/j.scitotenv.2023.168160

**Billen, G., Gu, B., Zhang, X., Garnier, J., Gimeno, B., Hayashi, K., Le Noe, J., Quemada, M., Sanz-Cobeña, A., Grinsven, H., de Vries, W., Lassaletta, L.** (forthcoming). Establishing system boundaries for estimating NUE at different scales. In: L. Lassaletta & A. Sanz-Cobeña, eds. *Guidance Document on nitrogen use efficiency indicators across multiple scales*. INMS (International Nitrogen Management System) Guidance Document Series. Edinburgh, UK Centre for Ecology & Hydrology.

**Birkhofer, K., Bezemer, T.M., Bloem, J., Bonkowski, M., Christensen, S., Dubois, D., Ekelund, F. *et al.*** 2008. Long-term organic farming fosters below and aboveground biota: Implications for soil quality, biological control and productivity. *Special Section: Enzymes in the Environment*, 40(9): 2297–2308. https://doi.org/10.1016/j.soilbio.2008.05.007

**Bittman, S., Dedina, M., Howard, C., Oenema, O. & Sutton, M.** 2014. *Options for ammonia mitigation: Guidance from the UNECE task force on reactive nitrogen*. Edinburgh, Centre for Ecology & Hydrology.

**Bittman, S., Worth, D., Hunt, D., Spiegal, S., Kleinman, P., Nanayakkara, S., Vendramini, J. *et al.*** 2023. Distribution of livestock sectors in Canada: Implications for manureshed management. *Journal of Environmental Quality*, 52(3): 596–609. https://doi.org/10.1002/jeq2.20457

**Bodirsky, B.L. & Müller, C.** 2014. Robust relationship between yields and nitrogen inputs indicates three ways to reduce nitrogen pollution. *Environmental Research Letters*, 9(11): 111005. http://dx.doi.org/10.1088/1748-9326/9/11/111005

**Bodirsky, B.L., Chen, D.M.-C., Weindl, I., Sörgel, B., Beier, F., Molina Bacca, E.J., Gaupp, F., Popp, A. & Lotze-Campen, H.** 2022. Integrating degrowth and efficiency perspectives enables an emission-neutral food system by 2100. *Nature Food*, 3(5): 341–348. https://doi.org/10.1038/s43016-022-00500-3

**Bogard, M.J., Del Giorgio, P.A., Boutet, L., Chaves, M.C.G., Prairie, Y.T., Merante, A. & Derry, A.M.** 2014. Oxic water column methanogenesis as a major component of aquatic $CH_4$ fluxes. *Nature Communications*, 5(1): 5350. https://doi.org/10.1038/ncomms6350

**Boumans, I.J.M.M., Schop, M., Bracke, M.B.M., de Boer, I.J.M., Gerrits, W.J.J. & Bokkers, E.A.M.** 2022. Feeding food losses and waste to pigs and poultry: Implications for feed quality and production. *Journal of Cleaner Production*, 378: 134623. https://doi.org/10.1016/j.jclepro.2022.134623

**Bouwman, L., Goldewijk, K.K., van der Hoek, K.W., Beusen, A.H.W., van Vuuren, D.P., Willems, J., Rufino, M.C. & Stehfest, E.** 2013. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. *Proceedings of the National Academy of Sciences*, 110(52): 20882–20887. https://doi.org/10.1073/pnas.1012878108

**Bouwman, A., Beusen, A., Lassaletta, L., van Apeldoorn, D., van Grinsven, H., Zhang, J. & van Ittersum, M.** 2017. Lessons from temporal and spatial patterns in global use of N and P fertilizer on cropland. *Scientific reports*, 7(1): 40366. http://dx.doi.org/10.1038/srep40366

**Bowman, W.D., Cleveland, C.C., Halada, Ĺ., Hreško, J. & Baron, J.S.** 2008. Negative impact of nitrogen deposition on soil buffering capacity. *Nature Geoscience*, 1(11): 767–770. https://doi.org/10.1038/ngeo339

**Boyle, L., Olmos, G., Llamas Moya, S., Palmer, M., Gleeson, D.E., O'Brien, B., Horan, B. *et al.*** 2008. *Cow welfare in grass based milk production systems*. Teagasc.

**Breitburg, D., Levin, L.A., Oschlies, A., Grégoire, M., Chavez, F.P., Conley, D.J., Garçon, V. *et al.*** 2018. Declining oxygen in the global ocean and coastal waters. *Science*, 359(6371): eaam7240. https://doi.org/10.1126/science.aam7240

**Brink, M., Janssens, G.P., Demeyer, P., Bağci, Ö. & Delezie, E.** 2022. Reduction of dietary crude protein and feed form: Impact on broiler litter quality, ammonia concentrations, excreta composition, performance, welfare and meat quality. *Animal Nutrition*, 9: 291–303. https://doi.org/10.1016/j.aninu.2021.12.009

**Brocławik, O., Łukawska-Matuszewska, K., Brodecka-Goluch, A. & Bolałek, J.** 2020. Impact of methane occurrence on iron speciation in the sediments of the Gdansk Basin (Southern Baltic Sea). *Science of the Total Environment*, 721: 137718. https://doi.org/10.1016/j.scitotenv.2020.137718

**Bruulsema, T.** 2018. Managing nutrients to mitigate soil pollution. *Environmental pollution*, 243: 1602–1605. https://doi.org/10.1016/j.envpol.2018.09.132

**Burg, V., Bowman, G., Haubensak, M., Baier, U. & Thees, O.** 2018. Valorization of an untapped resource: Energy and greenhouse gas emissions benefits of converting manure to biogas through anaerobic digestion. *Resources, Conservation and Recycling*, 136: 53–62. https://doi.org/10.1016/j.resconrec.2018.04.004

**Burkholder, J.M., Tomasko, D.A. & Touchette, B.W.** 2007. Seagrasses and eutrophication. *Journal of Experimental Marine Biology and Ecology*, 350(1–2): 46–72. https://doi.org/10.1016/j.jembe.2007.06.024

**Burlacot, A., Richaud, P., Gosset, A., Li-Beisson, Y. & Peltier, G.** 2020. Algal photosynthesis converts nitric oxide into nitrous oxide. *Proceedings of the National Academy of Sciences*, 117(5): 2704–2709. https://doi.org/10.1073/pnas.1915276117

**Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R. & Zechmeister-Boltenstern, S.** 2013. Nitrous oxide emissions from soils: How well do we understand the processes and their controls? *Philosophical Transactions of the Royal Society B: Biological Sciences*, 368(1621): 20130122. https://doi.org/10.1098/rstb.2013.0122

**Camargo, J.A. & Alonso, A.** 2006. Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. *Environment International*, 32(6): 831–849. https://doi.org/10.1016/j.envint.2006.05.002

**Campanati, C., Willer, D., Schubert, J. & Aldridge, D.C.** 2022. Sustainable intensification of aquaculture through nutrient recycling and circular economies: More fish, less waste, blue growth. *Reviews in Fisheries Science & Aquaculture*, 30(2): 143–169. https://doi.org/10.1080/23308249.2021.1897520

**Carbonell, V., Merbold, L., Díaz-Pinés, E., Dowling, T.P. & Butterbach-Bahl, K.** 2021. Nitrogen cycling in pastoral livestock systems in Sub-Saharan Africa: Knowns and unknowns. *Ecological Applications*, 31(6): e02368. https://doi.org/10.1002/eap.2368

**Cardenas, L., Bhogal, A., Chadwick, D., McGeough, K., Misselbrook, T., Rees, R., Thorman, R. *et al.*** 2019. Nitrogen use efficiency and nitrous oxide emissions from five UK fertilised grasslands. *Science of the Total Environment*, 661: 696–710. https://doi.org/10.1016/j.scitotenv.2019.01.082

**Cassman, K.G. & Dobermann, A.** 2022. Nitrogen and the future of agriculture: 20 years on. *Ambio*, 51(1): 17–24. https://doi.org/10.1007/s13280-021-01526-w

**Castillo, J., Kirk, G.J.D., Rivero, M.J. & Haefele, S.M.** 2023. Regional differences in nitrogen balance and nitrogen use efficiency in the rice-livestock system of Uruguay. *Frontiers in Sustainable Food Systems*, 7: 1104229. https://doi.org/10.3389/fsufs.2023.1104229

**CBD (Convention on Biological Diversity)**. 2022. CBD/COP/DEC/15/4. Decision adopted by The Conference of the Parties to the Convention on Biological Diversity. 15/4. Kunming-Montreal Global Biodiversity Framework. Convention on Biological Diversity. [Cited 5 May 2024]. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf

**Chatzimpiros, P. & Harchaoui, S.** 2023. Sevenfold variation in global feeding capacity depends on diets, land use and nitrogen management. *Nature Food*, 4(5): 372–383. https://doi.org/10.1038/s43016-023-00741-w

**Chen, J., Jin, C., Sun, S., Yang, D., He, Y., Gan, P., Nalume, W.G. *et al.*** 2023. Recognizing the challenges of composting: Critical strategies for control, recycling, and valorization of nitrogen loss. *Resources, Conservation and Recycling*, 198: 107172. https://doi.org/10.1016/j.resconrec.2023.107172

**Cherry, K., Mooney, S.J., Ramsden, S. & Shepherd, M.A.** 2012. Using field and farm nitrogen budgets to assess the effectiveness of actions mitigating N loss to water. *Agriculture, Ecosystems & Environment*, 147: 82–88. https://doi.org/10.1016/j.agee.2011.06.021

**Chipperfield, M. P. & Bekki, S.** 2024. Opinion: Stratospheric ozone – depletion, recovery and new challenges. *Atmospheric Chemistry and Physics*, 24(4): 2783–2802. https://doi.org/10.5194/acp-24-2783-2024

**Chowdhury, M., Wilkinson, R. & Sinclair, L.** 2023. Feeding lower-protein diets based on red clover and grass or alfalfa and corn silage does not affect milk production but improves nitrogen use efficiency in dairy cows. *Journal of Dairy Science*, 106(3): 1773–1789. https://doi.org/10.3168/jds.2022-22607

**Codispoti, L.A.** 2010. Interesting times for marine $N_2O$. *Science*, 327(5971): 1339–1340. https://doi.org/10.1126/science.1184945

**Cole, J.J., Pace, M.L., Carpenter, S.R. & Kitchell, J.F.** 2000. Persistence of net heterotrophy in lakes during nutrient addition and food web manipulations. *Limnology and Oceanography*, 45(8): 1718–1730. https://doi.org/10.4319/lo.2000.45.8.1718

**Compton, J., Pearlstein, S.L., Erban, L., Coulombe, R., Hatteberg, B., Henning, A., Brooks, J.R. & Selker, J.** 2021. Nitrogen inputs best predict farm field nitrate leaching in the Willamette Valley, Oregon. *Nutrient Cycling in Agroecosystems*, 120(2): 223–242. https://doi.org/10.1007/s10705-021-10145-6

**CONAMA (Conselho Nacional do Meio Ambiente)**. 2020. CONAMA Resolution No. 498/2020. CONAMA. https://conama.mma.gov.br/index.php?option=com_sisconama&view=atonormativo&id=726

**Conant, R.T., Paustian, K. & Elliott, E.T.** 2001. Grassland management and conversion into grassland: effects on soil carbon. *Ecological Applications*, 11(2): 343–355. https://doi.org/10.1890/1051-0761(2001)011[0343:GMACIG]2.0.CO;2

**Constantin, J., Beaudoin, N., Launay, M., Duval, J. & Mary, B.** 2012. Long-term nitrogen dynamics in various catch crop scenarios: Test and simulations with STICS model in a temperate climate. *Agriculture, Ecosystems & Environment*, 147: 36–46. https://doi.org/10.1016/j.agee.2011.06.006

**Cooney, R., de Sousa, D.B., Fernández-Ríos, A., Mellett, S., Rowan, N., Morse, A.P., Hayes, M. *et al.*** 2023. A circular economy framework for seafood waste valorisation to meet challenges and opportunities for intensive production and sustainability. *Journal of Cleaner Production*, 392: 136283. https://doi.org/10.1016/j.jclepro.2023.136283

**Corrado, S., Caldeira, C., Carmona-Garcia, G., Körner, I., Leip, A. & Sala, S.** 2020. Unveiling the potential for an efficient use of nitrogen along the food supply and consumption chain. *Global Food Security*, 25: 100368. https://doi.org/10.1016/j.gfs.2020.100368

**Cosby, B.J., Webb, R.R., Galloway J.N. & Deviney, N.A.** 2006. Acidic deposition impacts on natural resources in Shenandoah National Park. Technical report. NPS/NER/NRTR-2006/066. Philadelphia, National Park Service. https://npshistory.com/publications/shen/nrtr-2006-066.pdf

**Cowley, F., Jennings, J., Cole, A. & Beauchemin, K.** 2019. Recent advances to improve nitrogen efficiency of grain-finishing cattle in North American and Australian feedlots. *Animal Production Science*, 59(11): 2082–2092. https://doi.org/10.1071/AN19259

**Craine, J.M., Elmore, A. & Angerer, J.P.** 2017. Long-term declines in dietary nutritional quality for North American cattle. *Environmental Research Letters*, 12(4): 044019. https://doi.org/10.1088/1748-9326/aa67a4

**Dal Molin, S.J., Ernani, P.R. & Gerber, J.M.** 2020. Soil acidification and nitrogen release following application of nitrogen fertilizers. *Communications in Soil Science and Plant Analysis*, 51(20): 2551–2558. https://doi.org/10.1080/00103624.2020.1845347

**Dao, H.T., Sharma, N.K., Swick, R.A. & Moss, A.F.** 2023. Feeding recycled food waste improved feed efficiency in laying hens from 24 to 43 weeks of age. *Scientific Reports*, 13(1): 8261. https://doi.org/10.1038/s41598-023-34878-2

**Das, A., Mishra, R., Rani, K., Kundu, S., Jayaraman, S. & Ch, S.** 2021. Improving nutrient use efficiency: Research, technology and policy. In: Srinivasarao, Ch., Balakrishnan, M., Krishnan, P., Sumanth Kumar,V.V. (Eds). 2021. Agricultural Research, Technology and Policy: Innovations and Advances, *ICAR – National Academy of Agricultural Research Management (NAARM),* Hyderabad, Telangana, India, pp. 191–227. www.naarm.org.in

**Davidson, E.A. & Janssens, I.A.** 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. *Nature*, 440(7081): 165–173. https://doi.org/10.1038/nature04514

**Davidson, E.A. & Kanter, D.** 2014. Inventories and scenarios of nitrous oxide emissions. *Environmental Research Letters*, 9(10): 105012. http://dx.doi.org/10.1088/1748-9326/9/10/105012

**Davies, C.A., Robertson, A.D. & McNamara, N.P.** 2021. The importance of nitrogen for net carbon sequestration when considering natural climate solutions. *Global Change Biology*, 27(2): 218–219. https://doi.org/10.1111/gcb.15381

**de Boer, I.J.M. & van Ittersum, M.K.** 2018. Circularity in agricultural production. *Wageningen University & Research*: 74p. https://edepot.wur.nl/470625

**de Boer, H.C., van Mullekom, M. & Smolders, A.J.P.** 2024. Lower nitrate leaching from dairy cattle slurry compared to synthetic fertilizer calcium ammonium nitrate applied to grassland. *Environmental Pollution*, 344: 123088. https://doi.org/10.1016/j.envpol.2023.123088

**de Faccio Carvalho, P., Barro, R., Neto, A., De Albuquerque Nunes, P.A., De Moraes, A., Anghinoni, I., Bredemeier, C. *et al.*** 2018. Integrating the pastoral component in agricultural systems. *Revista Brasileira de Zootecnia*, 47. https://doi.org/10.1590/rbz4720170001

**de Vries, W., Kros, H., Reinds, G.J., Wamelink, W., Mol, J., van Dobben, H., Bobbink, R. *et al.*** 2007. Developments in deriving critical limits and modelling critical loads of nitrogen for terrestrial ecosystems in Europe. Report 1382. Wageningen, the Kingdom of the Netherlands., Alterra Wageningen University & Research. https://edepot.wur.nl/38943

**de Vries, W., Leip, A., Reinds, G.J., Kros, J., Lesschen, J.P. & Bouwman, A.** 2011. Comparison of land nitrogen budgets for European agriculture by various modelling approaches. *Environmental Pollution*, 159(11): 3254–3268. https://doi.org/10.1016/j.envpol.2011.03.038

**de Vries, W., Posch, M., Simpson, D., de Leeuw, F.A., van Grinsven, H.J., Schulte-Uebbing, L.F., Sutton, M.A. & Ros, G.H.** 2024. Trends and geographic variation in adverse impacts of nitrogen use in Europe on human health, climate, and ecosystems: A review. *Earth-Science Reviews*: 104789. https://doi.org/10.1016/j.earscirev.2024.104789

**Delgado, J.A. & Follett, R.F.** 2011. Advances in nitrogen management for water quality. *Journal of Soil and Water Conservation*, 66(1): 25A-26A. https://doi.org/10.2489/jswc.66.1.25A

**Derwent, R.G., Dollard, G.J. & Metcalfe, S.E.** 1988. On the nitrogen budget for the United Kingdom and north-west Europe. *Quarterly Journal of the Royal Meteorological Society*, 114(482): 1127–1152. https://doi.org/10.1002/qj.49711448212

**Deutzmann, J.S., Stief, P., Brandes, J. & Schink, B.** 2014. Anaerobic methane oxidation coupled to denitrification is the dominant methane sink in a deep lake. *Proceedings of the National Academy of Sciences*, 111(51): 18273–18278. https://doi.org/10.1073/pnas.1411617111

**Devkota, M., Frija, A., Dhehibi, B., Rudiger, U., Alary, V., M'hamed, H.C., Louahdi, N., Idoudi, Z. & Rekik, M.** 2022. Better crop-livestock integration for enhanced agricultural system resilience and food security in the changing climate: Case study from low-rainfall areas of North Africa. In: M. Behnassi, M.B. Baig, M.T. Sraïri, A.A. Alsheikh & A.W.A. Abu Risheh, eds. *Food Security and Climate-Smart Food Systems: Building Resilience for the Global South*. Switzerland, Springer Cham. https://link.springer.com/book/10.1007/978-3-030-92738-7

**Dimkpa, C.O., Fugice, J., Singh, U. & Lewis, T.D.** 2020. Development of fertilizers for enhanced nitrogen use efficiency – Trends and perspectives. *Science of The Total Environment*, 731: 139113. https://doi.org/10.1016/j.scitotenv.2020.139113

**Dise, N.B., Ashmore, M., Belyazid, S., Bleeker, A., Bobbink, R., de Vries, W., Erisman, J.W. *et al.*** 2011. Nitrogen as a threat to European terrestrial biodiversity. In: M.A. Sutton, C.M. Howard, J.W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. van Grinsven & B. Grizzetti, eds. *The European Nitrogen Assessment*. First edition, pp. 463–494. Cambridge University Press. https://doi.org/10.1017/CBO9780511976988.023

**Do Rosário Gomes, H., Goes, J.I., Matondkar, S.G.P., Buskey, E.J., Basu, S., Parab, S. & Thoppil, P.** 2014. Massive outbreaks of Noctiluca scintillans blooms in the Arabian Sea due to spread of hypoxia. *Nature Communications*, 5(1): 4862. https://doi.org/10.1038/ncomms5862

**Dobermann, A.** 2006. Invited paper: Nitrogen use efficiency in cereal systems. *Proceedings of the 13th Australian Agronomy Conference.* https://www.cabidigitallibrary.org/doi/pdf/10.5555/20193228440

**Dobermann, A., Bruulsema, T., Cakmak, I., Gerard, B., Majumdar, K., McLaughlin, M., Reidsma, P. *et al.*** 2022. Responsible plant nutrition: A new paradigm to support food system transformation. *Global Food Security*, 33: 100636. https://doi.org/10.1016/j.gfs.2022.100636

**Doney, S.C., Fabry, V.J., Feely, R.A. & Kleypas, J.A.** 2009. Ocean Acidification: The Other $CO_2$ Problem. *Annual Review of Marine Science*, 1(1): 169–192. https://doi.org/10.1146/annurev.marine.010908.163834

**Dos Santos Cordeiro, C.F., Lopes, B.P., Batista, G.D., Araujo, F.F., Tiritan, C.S. & Echer, F.R.** 2021. Inoculation and nitrogen fertilization improve nitrogen soil stock and nutrition to soybeans in degraded pastures with sandy soil. *Communications in Soil Science and Plant Analysis*, 52(12): 1388–1398. https://doi.org/10.1080/00103624.2021.1885685

**Dou, Z., Toth, J.D. & Westendorf, M.L.** 2018. Food waste for livestock feeding: Feasibility, safety, and sustainability implications. *Global Food Security*, 17: 154–161. https://doi.org/10.1016/j.gfs.2017.12.003

**Drechsel, P., Heffer, P., Magen, H., Mikkelsen, R. & Wichelns, D.** (Eds). 2015. Managing water and fertilizer for sustainable agricultural intensification. International Fertilizer Industry Association (IFA), International Water Management Institute (IWMI), International Plant Nutrition Institute (IPNI), and International Potash Institute (IPI). [Cited 15 December 2023] https://www.iwmi.cgiar.org/Publications/Books/PDF/managing_water_and_fertilizer_for_sustainable_agricultural_intensification.pdf

**Durand, P., Breuer, L., Johnes, P.J., Billen, G., Butturini, A., Pinay, G., van Grinsven, H. *et al.*** 2011. Nitrogen processes in aquatic ecosystems. In: A. Bleeker, B. Grizzetti, C.M. Howard, G. Billen, H. van Grinsven, J.W. Erisman, M.A. Sutton & P. Grennfelt, eds. *The European Nitrogen Assessment: Sources, Effects and Policy Perspectives*. pp. 126–146. Cambridge, Cambridge University Press. https://doi.org/10.1017/CBO9780511976988.010

**Earl, S.R., Valett, H.M. & Webster, J.R.** 2006. Nitrogen saturation in stream ecosystems. *Ecology*, 87(12): 3140–3151. https://doi.org/10.1890/0012-9658(2006)87[3140:NSISE]2.0.CO;2

**Ebi, K.L. & Loladze, I.** 2019. Elevated atmospheric $CO_2$ concentrations and climate change will affect our food's quality and quantity. *The Lancet Planetary Health*, 3(7): e283–e284. http://dx.doi.org/10.1016/S2542-5196(19)30108-1

**Einarsson, R.** 2024. *Nitrogen in the food system*. TABLE Explainer. TABLE, University of Oxford, Swedish University of Agricultural Sciences, and Wageningen University and Research. https://doi.org/10.56661/2fa45626

**Einarsson, R. & Cederberg, C.** 2019. Is the nitrogen footprint fit for purpose? An assessment of models and proposed uses. *Journal of Environmental Management*, 240: 198–208. https://doi.org/10.1016/j.jenvman.2019.03.083

**Ellen MacArthur Foundation**. 2019. The circular economy in detail. In: *The circular economy in detail*. [Cited 16 April 2024]. https://www.ellenmacarthurfoundation.org/the-circular-economy-in-detail-deep-dive

**Elliott, J.A., Jones, I.D. & Thackeray, S.J.** 2006. Testing the sensitivity of phytoplankton communities to changes in water temperature and nutrient load in a temperate lake. *Hydrobiologia*, 559(1): 401–411. https://doi.org/10.1007/s10750-005-1233-y

**El-Sheekh, M., Abdel-Daim, M.M., Okba, M., Gharib, S., Soliman, A. & El-Kassas, H.** 2021. Green technology for bioremediation of the eutrophication phenomenon in aquatic ecosystems: a review. *African Journal of Aquatic Science*, 46(3): 274–292. https://doi.org/10.2989/16085914.2020.1860892

**Erisman, J.W., Sutton, M.A., Galloway, J., Klimont, Z. & Winiwarter, W.** 2008. How a century of ammonia synthesis changed the world. *Nature Geoscience*, 1(10): 636–639. https://doi.org/10.1038/ngeo325

**Erisman, J.W., Leach, A., Bleeker, A., Atwell, B., Cattaneo, L. & Galloway, J.** 2018. An integrated approach to a nitrogen use efficiency (NUE) indicator for the food production-consumption chain. *Sustainability*, 10(4): 925. https://doi.org/10.3390/su10040925

**EUNEP (European Nitrogen Expert Panel)**. 2015. Nitrogen use efficiency (NUE) – an indicator for the utilization of nitrogen in food systems. *Wageningen University, Alterra, Wageningen, the Kingdom of the Netherlands*. [Cited 15 October 2023]. https://www.eunep.com/wp-content/uploads/2023/12/Report-NUE-Indicator-Nitrogen-Expert-Panel-18-12-2015.pdf

**European Commission**. 2018. Report on the implementation by Member States of Directive 2009/38/EC on the establishment of a European Works Council or a procedure in community-scale undertakings and community-scale groups of undertakings for the purposes of informing and consulting employees (Recast). European Commission. [Cited 19 April 2024]. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A52018SC0187

**European Commission**. Undated. Bio-based products and processes. [Cited 5 February 2024]. https://research-and-innovation.ec.europa.eu/research-area/environment/bioeconomy/bio-based-products-and-processes_en

**Famiglietti, J.S.** 2014. The global groundwater crisis. *Nature Climate Change*, 4(11): 945–948. https://doi.org/10.1038/nclimate2425

**Fan, D. & Yang, F.** 2024. Assessing the impact of China's agricultural subsidy reform on fertilizer management: a county-level empirical analysis based on difference-in-difference model. *Frontiers in Sustainable Food Systems*, 7. https://www.frontiersin.org/articles/10.3389/fsufs.2023.1298425

**Fan, P., Mishra, A.K., Feng, S. & Su, M.** 2023. The effect of agricultural subsidies on chemical fertilizer use: Evidence from a new policy in China. *Journal of Environmental Management*, 344: 118423. https://doi.org/10.1016/j.jenvman.2023.118423

**FAO (Food and Agriculture Organization of the United Nations).** 2011. *Global Food Losses and Food Waste – Extent, causes and prevention*. Rome. https://www.fao.org/4/mb060e/mb060e00.pdf

**FAO**. 2015. *Food Losses and Waste in Latin America and the Caribbean*. Rome. https://openknowledge.fao.org/server/api/core/bitstreams/daa2a3ee-28b0-4b42-8a14-d9a993c26312/content

**FAO**. 2016. *Soils and Pulses: Symbiosis for life*. Rome. https://www.fao.org/3/i6437e/i6437e.pdf

**FAO**. 2017. *Voluntary Guidelines for Sustainable Soil Management*. Rome, Food and Agriculture Organization of the United Nations. https://www.fao.org/3/bl813e/bl813e.pdf

**FAO**. 2018a. *Nutrient flows and associated environmental impacts in livestock supply chains. Guidelines for assessment (Version 1)*. Livestock Environmental Assessment and Performance (LEAP) Partnership. Rome, FAO. https://www.fao.org/3/CA1328EN/ca1328en.pdf

**FAO.** 2018b. FAOSTAT Emission Shares dataset. [Cited 4 March 2024]. https://www.fao.org/faostat/en/#data/EM/visualize

**FAO.** 2018c. The 10 elements of agroecology. Guiding the transition to sustainable food and agricultural systems. Rome. https://openknowledge.fao.org/server/api/core/bitstreams/3d7778b3-8fba-4a32-8d13-f21dd5ef31cf/content

**FAO.** 2019. *The International Code of Conduct for the Sustainable Use and Management of Fertilizers*. Rome, FAO. https://doi.org/10.4060/CA5253EN

**FAO.** 2021a. *Pastoralism – Making variability work*. Rome, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cb5855en

**FAO.** 2021b. Aspirational principles and criteria for a sustainable bioeconomy. Rome, FAO. https://www.fao.org/3/cb3706en/cb3706en.pdf

**FAO.** 2022a. *Tackling food loss and waste: A triple win opportunity*. [Cited 2 October 2024]. https://www.unep.org/news-and-stories/press-release/tackling-food-loss-and-waste-triple-win-opportunity-fao-unep

**FAO.** 2022b. Establishing residue supply chains to reduce open burning. The case of rice straw and renewable energy in Punjab, India. Rome, Food and Agriculture Organization of the United Nations. https://doi. org/10.4060/cb9570en

**FAO.** 2022c. Voluntary code of conduct for food loss and waste reduction. Rome, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cb9433en

**FAO.** 2022d. *Tracking progress on food and agriculture-related SDG indicators 2022*. Rome. FAO. https://doi.org/10.4060/cc1403en

**FAO.** 2023a. Cropland Nutrient Balance. In: *FAOSTAT*. [Cited 22 September 2024]. https://www.fao.org/faostat/en/#data/ESB

**FAO.** 2023b. *Cropland nutrient balance – Global, regional and country trends, 1961–2021. FAOSTAT Analytical Briefs, No. 74.* https://doi.org/10.4060/cc8962en

**FAO.** 2023c. *Food safety implications from the use of environmental inhibitors in agrifood systems*. *Food Safety and Quality Series* No. 24. Rome, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cc8647en

**FAO.** 2023d. Achieving SDG 2 without breaching the 1.5 °C threshold: A global roadmap, Part 1. Rome, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cc9113en

**FAO.** 2023e. *Soils, where food begins: How can soils continue to sustain the growing need for food production in the current fertilizer crisis?* ITPS Soil letters #6. Intergovernmental Technical Panel on Soils. https://openknowledge.fao.org/server/api/core/bitstreams/a2952c41-05ff-4720-8a9a-44065dc440ce/content

**FAO.** 2024a. *The state of world fisheries and aquaculture 2024 – Blue transformation in action*. *The State of World Fisheries and Aquaculture* (SOFIA). Rome, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cd0683en

**FAO.** 2024b. The organic waste problem. [Cited 5 March 2024]. https://www.fao.org/science-technology-and-innovation/resources/stories/detail/economic-and-environmental-benefits-from-using-black-soldier-fly-larvae-to-digest-organic-waste/en

**FAO.** 2024c. Statistical Databases. In: *FAOSTAT*. [Cited 10 March 2024]. https://www.fao.org/faostat/en/#home

**FAO & ITPS (Intergovernmental Technical Panel on Soils).** 2015. *Status of the World's Soil Resources (SWSR) – Main Report. Food and Agriculture Organization of the United Nations and Intergovernmental Technical Panel on Soils*. Rome. https://www.fao.org/3/i5199e/I5199E.pdf

**FAO & NZAGRC (New Zealand Agricultural Greenhouse Gas Research Centre)**. 2019. *Options for low emission development in the Uganda dairy sector – reducing enteric methane for food security and livelihoods*. Rome. https://www.fao.org/3/ca3375en/ca3375en.pdf

**FAO & UNECE (United Nations Economic Commission for Europe)**. 2023. *Sustainable and circular bioeconomy in forest-based industries: How to get there*. Geneva Timber and Forest Discussion Paper 96. Geneva, Switzerland, United Nations and Food and Agriculture Organization of the United Nations. https://unece.org/sites/default/files/2023-11/ECE_TIM_2023_Inf.4_FAO_EFC_2023_Inf.4.pdf

**FAO, IOC (Intergovernmental Oceanographic Commission) & IAEA (nternational Atomic Energy Agency)**. 2023. *Joint technical guidance for the implementation of early warning systems for harmful algal blooms.* FAO Fisheries and Aquaculture Technical Paper No 690. Rome, FAO. https://doi.org/10.4060/cc4794en

**FAO, ITPS, GSBI (Global Soil Biodiversity Initiative), SCBD (Convention on Biological Diversity) & EC (European Commission)**. 2020. *State of Knowledge of Soil Biodiversity: Status, Challenges, and Potentialities, Report 2020*. Rome, FAO, ITPS, GSBI, SCBD & EC. https://www.fao.org/3/cb1928en/cb1928en.pdf

**Farias, G.D., Dubeux, J.C.B., Savian, J.V., Duarte, L.P., Martins, A.P., Tiecher, T., Alves, L.A., de Faccio Carvalho, P.C. & Bremm, C.** 2020. Integrated crop-livestock system with system fertilization approach improves food production and resource-use efficiency in agricultural lands. *Agronomy for Sustainable Development*, 40(6): 39. https://doi.org/10.1007/s13593-020-00643-2

**Farmaha, B.S., Eskridge, K.M., Cassman, K.G., Specht, J.E., Yang, H. & Grassini, P.** 2016. Rotation impact on on-farm yield and input-use efficiency in high-yield irrigated maize-soybean systems. *Agronomy Journal*, 108(6): 2313–2321. https://doi.org/10.2134/agronj2016.01.0046

**Fenetahun, Y., Xinwen, X. & Yong-dong, W.** 2018. Assessment of rangeland management approaches in Yabello: Implication for improved rangeland and pastoralist livelihoods. Review Paper. *International Journal of Advanced Research in Botany,* 4(3): 16–25.

**Ferreira, C.M., Soares, H.M. & Soares, E.V.** 2019. Promising bacterial genera for agricultural practices: An insight on plant growth-promoting properties and microbial safety aspects. *Science of the Total Environment*, 682: 779–799. https://doi.org/10.1016/j.scitotenv.2019.04.225

**Fixen, P.E.** 2020. A brief account of the genesis of 4R nutrient stewardship. *Agronomy Journal*, 112(5): 4511–4518. http://dx.doi.org/10.1002/agj2.20315

**Fixen, P., Brentrup, F., Bruulsema, T., Garcia, F., Norton, R. and Zingore, S**. 2015. Nutrient/Fertilizer Use Efficiency: Measurement, Current Situation and Trends. In: P. Drechsel, P. Heffer, H. Magen, R. Mikkelsen & D. Wichelns, eds. *Managing water and fertilizer for sustainable agricultural intensification*. Paris, International Fertilizer Industry Association (IFA), International Water Management Institute (IWMI), International Plant Nutrition Institute (IPNI), and International Potash Institute (IPI).

**Forster, P., Storelvmo, T., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D.J., *et al.*** 2021. The Earth's Energy Budget, Climate Feedbacks, and Climate Sensitivity. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, et al.., eds. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom and New York, NY, Cambridge University Press. http://doi.org/10.1017/9781009157896.009

**Fowler, D., Steadman, C.E., Stevenson, D., Coyle, M., Rees, R.M., Skiba, U.M., Sutton, M.A. *et al.*** 2015. Effects of global change during the 21st century on the nitrogen cycle. *Atmospheric Chemistry and Physics*, 15(24): 13849–13893. https://doi.org/10.5194/acp-15-13849-2015

**Frame, C.H., Lau, E., Nolan, E.J., Goepfert, T.J. & Lehmann, M.F.** 2017. Acidification enhances hybrid $N_2O$ production associated with aquatic ammonia-oxidizing microorganisms. *Frontiers in Microbiology*, 7. https://doi.org/10.3389/fmicb.2016.02104

**Frank, M. & Husted, S.** 2024. Is India's largest fertilizer manufacturer misleading farmers and society using dubious plant and soil science? *Plant and Soil*, 496(1): 257–267. https://doi.org/10.1007/s11104-023-06191-4

**Franke, A.C. & Kotzé, E.** 2022. High-density grazing in southern Africa: Inspiration by nature leads to conservation? *Outlook on Agriculture*, 51(1): 67–74. https://doi.org/10.1177/00307270221075060

**Franke, A., van den Brand, G., Vanlauwe, B. & Giller, K.** 2018. Sustainable intensification through rotations with grain legumes in Sub-Saharan Africa: A review. *Agriculture, Ecosystems & Environment*, 261: 172–185. https://doi.org/10.1016/j.agee.2017.09.029

**Fukagawa, N.K., Ziska, L.H.** 2019. Rice: Importance for Global Nutrition. *Journal of Nutritional Science and Vitaminoly*, 65: S2-S3. http://doi.org/10.3177/jnsv.65.S2

**Fulweiler, R.W., Emery, H.E., Heiss, E.M. & Berounsky, V.M.** 2011. Assessing the role of pH in determining water column nitrification rates in a coastal system. *Estuaries and Coasts*, 34(6): 1095–1102. https://doi.org/10.1007/s12237-011-9432-4

**Galindo, F.S., da Silva, E.C., Pagliari, P.H., Fernandes, G.C., Rodrigues, W.L., Biagini, A.L.C., Baratella, E.B. *et al.*** 2021. Nitrogen recovery from fertilizer and use efficiency response to Bradyrhizobium sp. and Azospirillum brasilense combined with N rates in cowpea-wheat crop sequence. *Applied Soil Ecology*, 157: 103764. http://dx.doi.org/10.1016/j.apsoil.2020.103764

**Gallager, J. & Baker, J.** 1990. *Adirondack Lakes Survey: An interpretive analysis of fish communities and water chemistry, 1984–1987*. Technical report. Ray Brook, NY, Adirondack Lakes Survey Corporation. https://doi.org/10.2172/6173689

**Galland, W., Piola, F., Burlet, A., Mathieu, C., Nardy, M., Poussineau, S., Blazère, L. *et al.*** 2019. Biological denitrification inhibition (BDI) in the field: A strategy to improve plant nutrition and growth. *Soil Biology and Biochemistry*, 136: 107513. https://doi.org/10.1016/j.soilbio.2019.06.009

**Galles, K., Ham, J., Westover, E., Stratton, J., Wagner, J., Engle, T. & Bryant, T.C.** 2011. Influence of reduced nitrogen diets on ammonia emissions from cattle feedlot pens. *Atmosphere*, 2(4): 655–670. http://dx.doi.org/10.3390/atmos2040655

**Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B. & Cosby, B.J.** 2003. The nitrogen cascade. *BioScience*, 53(4): 341. https://doi.org/10.1641/0006-3568(2003)053[0341:TNC]2.0.CO;2

**Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z., Freney, J.R., Martinelli, L.A., Seitzinger, S.P. & Sutton, M.A.** 2008. Transformation of the nitrogen cycle: Recent trends, questions and potential solutions. *Science*, 320(5878): 889–892. https://doi.org/10.1126/science.1136674

**Galloway, J., Dentener, F., Burke, M., Dumont, E., Bouwman, A., Kohn, R.A., Mooney, H.A., Seitzinger, S. & Kroeze, C.** 2010. The impact of animal production systems on the nitrogen cycle. In: H. Steinfeld, H.A. Mooney, F. Schneider & L.E. Neville, eds. *Livestock in a Changing Landscape: Drivers, Consequences, and Responses. Volume 1*. Washington, DC, Island Press.

**Galloway, J.N., Leach, A.M., Bleeker, A. & Erisman, J.W.** 2013. A chronology of human understanding of the nitrogen cycle. *Philosophical Transactions of the Royal Society of London. Series B, Biological sciences*, 368(1621): 20130120. https://doi.org/10.1098/rstb.2013.0120

**Ganeshamurthy, A.N.** 2009. *Annual Report 2008–09. IIHR, Bangalore*. Bangalore. [Cited 15 November 2023] https://www.iihr.res.in/sites/default/files/IIHR_Annual_Report_2008-09_0.pdf

**Gao, Y., Wang, J., Ge, Y., Lei, Y., Wei, X., Xu, Y. & Zheng, X.** 2024. Partial substitution of nitrogen fertilizers by organic products of rural waste co-composting impacts on farmland soil quality. *Environmental Technology & Innovation*, 33: 103470. https://doi.org/10.1016/j.eti.2023.103470

**Garbellini, L.R., Chrispim, M.C., Silveira, J.E. & Pacca, S.A.** 2023. (Eco)toxicological impact potential from inorganic substances in biosolids: Real data-based suggestions for regulatory improvements. *Environmental Nanotechnology, Monitoring & Management*, 20: 100846. https://doi.org/10.1016/j.enmm.2023.100846

**Garnier, J., Anglade, J., Benoit, M., Billen, G., Puech, T., Ramarson, A., Passy, P. *et al.*** 2016. Reconnecting crop and cattle farming to reduce nitrogen losses to river water of an intensive agricultural catchment (Seine basin, France): Past, present and future. *Environmental Science & Policy*, 63: 76–90. https://doi.org/10.1016/j.envsci.2016.04.019

**Garnier, J., Billen, G., Aguilera, E., Lassaletta, L., Einarsson, R., Serra, J., do Rosário Cameira, M., Marques-dos-Santos, C. & Sanz-Cobena, A.** 2023. How much can changes in the agro-food system reduce agricultural nitrogen losses to the environment? Example of a temperate-Mediterranean gradient. *Journal of Environmental Management*, 337: 117732. https://doi.org/10.1016/j.jenvman.2023.117732

**Gaspareto, R.N., Jalal, A., Ito, W.C., Oliveira, C.E., Garcia, C.M., Boleta, E.H., Rosa, P.A. *et al.*** 2023. Inoculation with plant growth-promoting bacteria and nitrogen doses improves wheat productivity and nitrogen use efficiency. *Microorganisms*, 11(4). https://doi.org/10.3390/microorganisms11041046

**Gaudin, A.C., Janovicek, K., Deen, B. & Hooker, D.C.** 2015a. Wheat improves nitrogen use efficiency of maize and soybean-based cropping systems. *Agriculture, Ecosystems & Environment*, 210: 1–10. https://doi.org/10.1016/j.agee.2015.04.034

**Gaudin, A.C., Tolhurst, T.N., Ker, A.P., Janovicek, K., Tortora, C., Martin, R.C. & Deen, W.** 2015b. Increasing crop diversity mitigates weather variations and improves yield stability. *PloS ONE*, 10(2): e0113261. https://doi.org/10.1371/journal.pone.0113261

**Gautam, M., Laborde, D., Mamun, A., Martin, W., Piñeiro, V. & Vos, R.** 2022. *Repurposing Agricultural Policies and Support: Options to Transform Agriculture and Food Systems to Better Serve the Health of People, Economies, and the Planet*. The World Bank and IFPRI (International Food Policy Research Institute). https://hdl.handle.net/10986/36875

**Geissdoerfer, M., Pieroni, M.P.P., Pigosso, D.C.A. & Soufani, K.** 2020. Circular business models: A review. *Journal of Cleaner Production*, 277: 123741. https://doi.org/10.1016/j.jclepro.2020.123741

**Geisseler, D. & Scow, K.M.** 2014. Long-term effects of mineral fertilizers on soil microorganisms – A review. *Soil Biology and Biochemistry*, 75: 54–63. https://doi.org/10.1016/j.soilbio.2014.03.023

**Gelardi, D.L., Rath, D. & Kruger, C.E.** 2023. Grounding United States policies and programmes in soil carbon science: Strengths, limitations, and opportunities. *Frontiers in Sustainable Food Systems*, 7. https://www.frontiersin.org/articles/10.3389/fsufs.2023.1188133

**Gerber, P., Vellinga, T.V. & Steinfeld, H.** 2010. Issues and options in addressing the environmental consequences of livestock sector's growth. *Meat Science*, 84(2): 244–247. https://doi.org/10.1016/j.meatsci.2009.10.016

**Gerber, P.J., Steinfeld, Henning, Henderson, B., Mottet, A., Opio, C.I., Dijkman, J., Falcucci, A. & Tempio, G.** 2013. *Tackling climate change through livestock: a global assessment of emissions and mitigation opportunities*. Rome, FAO. https://www.fao.org/4/i3437e/i3437e.pdf

**Gerber, P.J., Uwizeye, A., Schulte, R.P., Opio, C.I. & de Boer, I.** 2014. Nutrient use efficiency: a valuable approach to benchmark the sustainability of nutrient use in global livestock production? *Current Opinion in Environmental Sustainability*, 9: 122–130. https://doi.org/10.1016/j.cosust.2014.09.007

**Gerber, J.S., Ray, D.K., Makowski, D., Butler, E.E., Mueller, N.D., West, P.C., Johnson, J.A. *et al.*** 2024. Global spatially explicit yield gap time trends reveal regions at risk of future crop yield stagnation. *Nature Food*, 5: 125–135. http://dx.doi.org/10.1038/s43016-023-00913-8

**Gerten, D., Heck, V., Jägermeyr, J., Bodirsky, B.L., Fetzer, I., Jalava, M., Kummu, M. *et al.*** 2020. Feeding ten billion people is possible within four terrestrial planetary boundaries. *Nature Sustainability*, 3(3): 200–208. https://doi.org/10.1038/s41893-019-0465-1

**Giambalvo, D., Stringi, L., Durante, G., Amato, G. & Frenda, A.S.** 2004. Nitrogen efficiency component analysis in wheat under rainfed Mediterranean conditions: Effects of crop rotation and nitrogen fertilization. In: C. Cantero-Martínez C. & D. Gabiña D, eds. *Mediterranean Rainfed Agriculture: Strategies for Sustainability*. Zaragoza, Spain, Mediterranean Agronomic Institute of Zaragoza. http://om.ciheam.org/om/pdf/a60/04600059.pdf

**Gilbert, P.M., Harrison, J., Heil, C. & Seitzinger, S.** 2006. Escalating worldwide use of urea: A global change contributing to coastal eutrophication. *Biogeochemistry*, 77: 441–463. http://dx.doi.org/10.1007/s10533-005-3070-5

**Gomez San Juan, M., Bogdanski, A. & Dubois, O.** 2019. *Towards sustainable bioeconomy –Lessons learned from case studies*. Environment and Natural Resources Management Working Paper 73. Rome, FAO. https://openknowledge.fao.org/handle/20.500.14283/ca4352en

**Gomez San Juan, M., Harnett, S. & Albinelli, I.** 2022a. *Sustainable and circular bioeconomy in the climate agenda: Opportunities to transform agrifood systems*. Rome, FAO. https://doi.org/10.4060/cc2668en

**Gomez San Juan, M., Harnett, S. & Albinelli, I.** 2022b. *Sustainable and circular bioeconomy in the biodiversity agenda: Opportunities to conserve and restore biodiversity in agrifood systems through bioeconomy practices*. Rome, Food and Agriculture Organization of the United Nations. https://doi.org/10.4060/cc3417en

**Gomez San Juan, M.** 2024. The bioeconomy toolbox. A guide to support the development of sustainable bioeconomy strategies and policies. *Environment and Natural Resources Management Working Paper 99*. Rome, Food and Agriculture Organization of the United Nations. https://openknowledge.fao.org/handle/20.500.14283/cc8856en

**Grandy, A.S., Loecke, T.D., Parr, S. & Robertson, G.P.** 2006. Long-term trends in nitrous oxide emissions, soil nitrogen and crop yields of till and no-till cropping systems. *Journal of Environmental Quality*, 35(4): 1487–1495. https://doi.org/10.2134/jeq2005.0166

**Grasset, C., Sobek, S., Scharnweber, K., Moras, S., Villwock, H., Andersson, S., Hiller, C. *et al.*** 2020. The $CO_2$-equivalent balance of freshwater ecosystems is non-linearly related to productivity. *Global Change Biology*, 26(10): 5705–5715. https://doi.org/10.1111/gcb.15284

**Greaver, T.L., Clark, C.M., Compton, J.E., Vallano, D., Talhelm, A.F., Weaver, C.P., Band, L.E. *et al.*** 2016. Key ecological responses to nitrogen are altered by climate change. *Nature Climate Change*, 6(9): 836–843. https://doi.org/10.1038/nclimate3088

**Greff, B., Szigeti, J., Nagy, Á., Lakatos, E. & Varga, L.** 2022. Influence of microbial inoculants on co-composting of lignocellulosic crop residues with farm animal manure: A review. *Journal of Environmental Management*, 302: 114088. https://doi.org/10.1016/j.jenvman.2021.114088

**Griffith, A.W. & Gobler, C.J.** 2020. Harmful algal blooms: A climate change co-stressor in marine and freshwater ecosystems. *Harmful Algae*, 91: 101590. https://doi.org/10.1016/j.hal.2019.03.008

**Grizzetti, B., Bouraoui, F., Billen, G., van Grinsven, H., Cardoso, A.C., Thieu, V., Garnier, J. *et al.*** 2011. Nitrogen as a threat to European water quality. In: M.A. Sutton, C.M. Howard, J.W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. van Grinsven & B. Grizzetti, eds. *The European Nitrogen Assessment*. First edition, pp. 379–404. Cambridge University Press. https://doi.org/10.1017/CBO9780511976988.020

**Grizzetti, B., Pretato, U., Lassaletta, L., Billen, G. & Garnier, J.** 2013. The contribution of food waste to global and European nitrogen pollution. *Environmental Science & Policy*, 33: 186–195. https://doi.org/10.1016/j.envsci.2013.05.013

**Groenestein, C.M., Hutchings, N.J., Haenel, H.D., Amon, B., Menzi, H., Mikkelsen, M.H., Misselbrook, T.H. *et al.*** 2019. Comparison of ammonia emissions related to nitrogen use efficiency of livestock production in Europe. *Journal of Cleaner Production*, 211: 1162–1170. https://doi.org/10.1016/j.jclepro.2018.11.143

**Gu, B.** 2022. Recoupling livestock and crops. *Nature Food*, 3(2): 102–103. https://doi.org/10.1038/s43016-022-00466-2

**Gu, B., Zhang, X., Lam, S.K., Yu, Y., van Grinsven, H.J.M., Zhang, S., Wang, X. *et al.*** 2023. Cost-effective mitigation of nitrogen pollution from global croplands. *Nature*, 613(7942): 77–84. https://doi.org/10.1038/s41586-022-05481-8

**Guardia, G., Aguilera, E., Vallejo, A., Sanz-Cobena, A., Alonso-Ayuso, M. & Quemada, M.** 2019. Effective climate change mitigation through cover cropping and integrated fertilization: A global warming potential assessment from a 10-year field experiment. *Journal of Cleaner Production*, 241: 118307. https://doi.org/10.1016/j.jclepro.2019.118307

**Guardia, G., García-Gutiérrez, S., Rodríguez-Pérez, R., Recio, J. & Vallejo, A.** 2021. Increasing N use efficiency while decreasing gaseous N losses in a non-tilled wheat (Triticum aestivum L.) crop using a double inhibitor. *Agriculture, Ecosystems & Environment*, 319: 107546. https://doi.org/10.1016/j.agee.2021.107546

**Guèdègbé, T. & Doukkali, M.R.** 2018. *Fertilizer use in Africa: A price issue*. Policy Brief 18/27. OCP Policy Center. https://www.policycenter.ma/sites/default/files/2021-01/OCPPC-PB1827-ENG.pdf

**Guenet, B., Gabrielle, B., Chenu, C., Arrouays, D., Balesdent, J., Bernoux, M., Bruni, E. *et al.*** 2021. Can $N_2O$ emissions offset the benefits from soil organic carbon storage? *Global Change Biology*, 27(2): 237–256. https://doi.org/10.1111/gcb.15342

**Guo, J.H., Liu, X.J., Zhang, Y., Shen, J.L., Han, W.X., Zhang, W.F., Christie, P. *et al.*** 2010. Significant acidification in major Chinese croplands. *Science*, 327(5968): 1008–1010. https://doi.org/10.1126/science.1182570

**Hack, C.M., Porta, M., Schäufele, R. & Grimoldi, A.A.** 2019. Arbuscular mycorrhiza mediated effects on growth, mineral nutrition and biological nitrogen fixation of Melilotus alba Med. in a subtropical grassland soil. *Applied Soil Ecology*, 134: 38–44. https://doi.org/10.1016/j.apsoil.2018.10.008

**Halpern, B.S., Selkoe, K.A., Micheli, F. & Kappel, C.V.** 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. *Conservation Biology*, 21(5): 1301–1315. https://doi.org/10.1111/j.1523-1739.2007.00752.x

**Hamza, M. & Anderson, W.K.** 2005. Soil compaction in cropping systems: A review of the nature, causes and possible solutions. *Soil and Tillage Research*, 82(2): 121–145. https://doi.org/10.1016/j.still.2004.08.009

**Hansen, S., Berland Frøseth, R., Stenberg, M., Stalenga, J., Olesen, J.E., Krauss, M., Radzikowski, P. *et al.*** 2019. Reviews and syntheses: Review of causes and sources of $N_2O$ emissions and $NO_3$ leaching from organic arable crop rotations. *Biogeosciences*, 16(14): 2795–2819. https://doi.org/10.5194/bg-16-2795-2019

**Hautier, Y., Niklaus, P.A. & Hector, A.** 2009. Competition for light causes plant Bbiodiversity loss after eutrophication. *Science*, 324(5927): 636–638. https://doi.org/10.1126/science.1169640

**Hazell, P.B.** 2009. *The Asian Green Revolution*. Intl Food Policy Res Inst.

**Herrero, M., Thornton, P.K., Notenbaert, A.M., Wood, S., Msangi, S., Freeman, H., Bossio, D. *et al.*** 2010. Smart investments in sustainable food production: Revisiting mixed crop-livestock systems. *Science*, 327(5967): 822–825. https://doi.org/10.1126/science.1183725

**Hickman, J.E., Zingore, S., Galy-Lacaux, C., Kihara, J., Bekunda, M. & Palm, C.A.** 2020. Assessing synergies and trade-offs from nitrogen use in Africa | SpringerLink. [Cited 7 May 2024]. https://link.springer.com/chapter/10.1007/978-3-030-58065-0_5

**Holden, S. & Lunduka, R.** 2012. Do fertilizer subsidies crowd out organic manures? The case of Malawi. *Agricultural Economics*, 43(3): 303–314. https://doi.org/10.1111/j.1574-0862.2012.00584.x

**Houlton, B.Z., Almaraz, M., Aneja, V., Austin, A.T., Bai, E., Cassman, K.G., Compton, J.E. *et al.*** 2019. A world of cobenefits: Solving the global nitrogen challenge. *Earth's Future*, 7(8): 865–872. https://doi.org/10.1029/2019EF001222

**Howarth, R., Swaney, D., Billen, G., Garnier, J., Hong, B., Humborg, C., Johnes, P., Mörth, C.-M. & Marino, R.** 2012. Nitrogen fluxes from the landscape are controlled by net anthropogenic nitrogen inputs and by climate. *Frontiers in Ecology and the Environment*, 10(1): 37–43. https://doi.org/10.1890/100178

**Hristov, A., Heyler, K., Schurman, E., Griswold, K., Topper, P., Hile, M., Ishler, V., Fabian-Wheeler, E. & Dinh, S.** 2015. Case study: Reducing dietary protein decreased the ammonia emitting potential of manure from commercial dairy farms. *The Professional Animal Scientist*, 31(1): 68–79. http://dx.doi.org/10.15232/pas.2014-01360

**Huang, T., Yang, H., Huang, C. & Ju, X.** 2017. Effect of fertilizer N rates and straw management on yield-scaled nitrous oxide emissions in a maize-wheat double cropping system. *Field Crops Research*, 204: 1–11. https://doi.org/10.1016/j.fcr.2017.01.004

**Hutchings, N.J., Sørensen, P., Cordovil, C.M. d. S., Leip, A. & Amon, B.** 2020. Measures to increase the nitrogen use efficiency of European agricultural production. *Global Food Security*, 26:100381. https://doi.org/10.1016/j.gfs.2020.100381

**Huygens, D., Orveillon, G., Lugato, E., Tavazzi, S., Comero, S., Jones, A., Gawlik, B. & Saveyn, H.** 2020. *Technical proposals for the safe use of processed manure above the threshold established for Nitrate Vulnerable Zones by the Nitrates Directive (91/676/EEC)*. Luxembourg, Publications Office of the European Union. https://dx.doi.org/10.2760/373351

**IAEA & FAO**. 2008. *Guidelines for sustainable manure management in Asian livestock production systems*. Joint FAO/IAEA Division of Nuclear Techniques in Food and Agriculture. https://www-pub.iaea.org/MTCD/Publications/PDF/TE_1582_web.pdf

**Ibáñez, A., Garrido-Chamorro, S., Vasco-Cárdenas, M. & Barreiro, C.** 2023. From Lab to Field: Biofertilizers in the 21st Century. *Horticulturae*, 9(12): 1306. https://doi.org/10.3390/horticulturae9121306

**Ibrahim, W., Graham, M. & Leitner, S.** 2021. *Characterization of pig manure management and associated environmental and health issues in central Uganda.* ILRI Research Report 96. Nairobi, ILRI (International Livestock Research Institute). https://hdl.handle.net/10568/117268

**IFA (International Fertilizer Association)**. 2020. International Fertilizer Association Statistics. [Cited 15 July 2023]. https://www.ifastat.org/databases/plant-nutrition

**IHME (Institute for Health Metrics and Evaluation).** 2021. Air pollution. https://www.healthdata.org/research-analysis/health-risks-issues/air-pollution

**Independent Group of Scientists appointed by the Secretary-General**. 2019. *Global Sustainable Development Report 2019: The future is now – science for achieving sustainable development*. New York, United Nations. https://sustainabledevelopment.un.org/content/documents/24797GSDR_report_2019.pdf

**IMF (International Monetary Fund)**. 2015. India: 2015 Article IV Consultation-Staff Report. IMF. [Cited 13 February 2024] https://www.imf.org/external/pubs/ft/scr/2015/cr1561.pdf

**IPCC (Intergovernmental Panel on Climate Change)**. 2021. Summary for Policymakers. In: V. Masson-Delmotte, P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou, eds. *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK and New York, USA, Cambridge University Press. http://doi.org/10.1017/9781009157896.001

**IPCC**. 2022. Climate change 2022: *Impacts, adaptation and vulnerability. Contribution of Working Group II to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change* [H.-O. Pörtner, D.C. Roberts, M. Tignor, E.S. Poloczanska, K. Mintenbeck, A. Alegría, M. Craig, S. Langsdorf, S. Löschke, V. Möller, A. Okem, B. Rama (eds.)]. Cambridge and New York, Cambridge University Press. https://doi.org/10.1017/9781009325844

**IPCC**. 2023. Summary for Policymakers. In: H. Lee & J. Romero, eds. *Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. http://dx.doi.org/10.59327/IPCC/AR6-9789291691647

**Jacob, D.J. & Winner, D.A.** 2009. Effect of climate change on air quality. *Atmospheric Environment*, 43(1): 51–63. https://doi.org/10.1016/j.atmosenv.2008.09.051

**Jariwala, H., Santos, R.M., Lauzon, J.D., Dutta, A. & Wai Chiang, Y.** 2022. Controlled release fertilizers (CRFs) for climate-smart agriculture practices: A comprehensive review on release mechanism, materials, methods of preparation and effect on environmental parameters. *Environmental Science and Pollution Research*, 29: 53967–53995. http://dx.doi.org/10.1007/s11356-022-20890-y

**Jaurena, M., Durante, M., Devincenzi, T., Savian, J.V., Bendersky, D., Moojen, F.G., Pereira, M. *et al.*** 2021. Native grasslands at the core: A new paradigm of intensification for the campos of Southern South America to increase economic and environmental sustainability. *Frontiers in Sustainable Food Systems*, 5: 547834. https://doi.org/10.3389/fsufs.2021.547834

**Jayne, T.S., Mason, N.M., Burke, W.J. & Ariga, J.** 2018. Taking stock of Africa's second-generation agricultural input subsidy programmes. *Food policy*, 75: 1–14. https://doi.org/10.1016/j.foodpol.2018.01.003

**Ji, Y., Liu, H. & Shi, Y.** 2020. Will China's fertilizer use continue to decline? Evidence from LMDI analysis based on crops, regions and fertilizer types. *PLoS ONE*, 15(8): e0237234. https://doi.org/10.1371/journal.pone.0237234

**Jöhnk, K.D., Huisman, J., Sharples, J., Sommeijer, B., Visser, P.M. & Stroom, J.M.** 2008. Summer heatwaves promote blooms of harmful cyanobacteria. *Global Change Biology*, 14(3): 495–512. https://doi.org/10.1111/j.1365-2486.2007.01510.x

**Johnson, B. & Goldblatt, C.** 2015. The nitrogen budget of Earth. *Earth-Science Reviews*, 148: 150–173. https://doi.org/10.1016/j.earscirev.2015.05.006

**Joint WHO/FAO/UNU Expert Consultation**. 2007. Protein and amino acid requirements in human nutrition: Report of a joint FAO/WHO/UNU expert consultation. https://iris.who.int/handle/10665/43411

**Jones, L., Provins, A., Holland, M., Mills, G., Hayes, F., Emmett, B., Hall, J. *et al.*** 2014. A review and application of the evidence for nitrogen impacts on ecosystem services. *Ecosystem Services*, 7: 76–88. https://doi.org/10.1016/j.ecoser.2013.09.001

**Jones, J., Guardia, G., Bruulsema, T.W. & Vallejo, A.** (forthcoming). Ways to estimate NUE in cropping systems. In: L. Lassaletta & A. Sanz-Cobeña, eds. *Guidance Document on nitrogen use efficiency indicators across multiple scales*. INMS Guidance Document Series. Edinburgh, UK Centre for Ecology & Hydrology.

**Joshi, P. & Visvanathan, C.** 2019. Sustainable management practices of food waste in Asia: Technological and policy drivers. *Journal of Environmental Management*, 247: 538–550. https://doi.org/10.1016/j.jenvman.2019.06.079

**Jovarauskas, D., Steponavičius, D., Kemzūraitė, A., Zinkevičius, R. & Venslauskas, K.** 2021. Comparative analysis of the environmental impact of conventional and precision spring wheat fertilization under various meteorological conditions. *Journal of Environmental Management*, 296(1): 113150. https://doi.org/10.1016/j.jenvman.2021.113150

**Ju, X., Gu, B., Wu, Y. & Galloway, J.N.** 2016. Reducing China's fertilizer use by increasing farm size. *Global Environmental Change*, 41: 26–32. https://doi.org/10.1016/j.gloenvcha.2016.08.005

**Kanter, D.R. & Searchinger, T.D.** 2018. A technology-forcing approach to reduce nitrogen pollution. *Nature Sustainability*, 1(10): 544–552. http://dx.doi.org/10.1038/s41893-018-0143-8

**Kanter, D.R., Bartolini, F., Kugelberg, S., Leip, A., Oenema, O. & Uwizeye, A.** 2020a. Nitrogen pollution policy beyond the farm. *Nature Food*, 1(1): 27–32. https://doi.org/10.1038/s43016-019-0001-5

**Kanter, D.R., Chodos, O., Nordland, O., Rutigliano, M. & Winiwarter, W.** 2020b. Gaps and opportunities in nitrogen pollution policies around the world. *Nature Sustainability*, 3(11): 956–963. https://doi.org/10.1038/s41893-020-0577-7

**Kastowski, M., Hinderer, M. & Vecsei, A.** 2011. Long-term carbon burial in European lakes: Analysis and estimate. *Global Biogeochemical Cycles*, 25(3). https://doi.org/10.1029/2010GB003874

**Kasymov, U., Ring, I., Gonchigsumlaa, G., Dejid, N. & Drees, L.** 2023. Exploring complementarity among interdependent pastoral institutions in Mongolia. *Sustainability Science*, 18(1): 115–131. https://doi.org/10.1007/s11625-022-01198-9

**Kaye, J.P. & Quemada, M.** 2017. Using cover crops to mitigate and adapt to climate change. A review. *Agronomy for Sustainable Development*, 37(1): 4. https://doi.org/10.1007/s13593-016-0410-x

**Kebalo, L.F., Garnier, P., Gonod, L.V. & Houot, S.** 2024. Using bio-based fertilizer derived from peri-urban wastes affects soil properties and lettuce yield and quality. *Scientia Horticulturae*, 324: 112599. https://doi.org/10.1016/j.scienta.2023.112599

**Kendzior, J., Raffa, D.W. & Bogdanski, A.** 2022. *A review of the impacts of crop production on the soil microbiome*. Rome, FAO. https://openknowledge.fao.org/server/api/core/bitstreams/367e75ca-590a-4409-b6ed-5e9ecd1a60f6/content

**Kermagoret, C., Claudet, J., Derolez, V., Nugues, M.M., Ouisse, V., Quillien, N., Baulaz, Y. *et al.*** 2019. How does eutrophication impact bundles of ecosystem services in multiple coastal habitats using state-and-transition models? *Ocean & Coastal Management*, 174: 144–153. https://doi.org/10.1016/j.ocecoaman.2019.03.028

**Ketema, M. & Bauer, S.** 2011. Determinants of manure and fertilizer applications in eastern highlands of Ethiopia. *Quarterly Journal of International Agriculture*, 50: 237–252. http://dx.doi.org/10.22004/ag.econ.155533

**Khonje, M.G., Nyondo, C., Chilora, L., Mangisoni, J.H., Ricker-Gilbert, J. & Burke, W.J.** 2022. Exploring adoption effects of subsidies and soil fertility management in Malawi. *Journal of Agricultural Economics*, 73(3): 874–892. https://doi.org/10.1111/1477-9552.12486

**Kissinger, W.F., Koelsch, R.K., Erickson, G.E. & Klopfenstein, T.J.** 2007. Characteristics of manure harvested from beef cattle feedlots. *Applied Engineering in Agriculture*, 23(3): 357–365. http://dx.doi.org/10.13031/2013.22685

**Koenig, K.M., McGinn, S.M. & Beauchemin, K.A.** 2013. Ammonia emissions and performance of backgrounding and finishing beef feedlot cattle fed barley-based diets varying in dietary crude protein concentration and rumen degradability12. *Journal of Animal Science*, 91(5): 2278–2294. https://doi.org/10.2527/jas.2012-5651

**Koning, L., Evers, A. & Šebek, L.** 2021. *Praktijkimplementatie $CH_4$ en $NH_3$ reductie via voerspoor-praktijkrapport 2020: Voerstrategieën om de methaan-en ammoniakemissie te reduceren in de melkveehouderij*. Wageningen, Wageningen Livestock Research. https://doi.org/10.18174/560334

**Kopáček, J., Ulrich, K.-U., Hejzlar, J., Borovec, J. & Stuchlık, E.** 2001. Natural inactivation of phosphorus by aluminum in atmospherically acidified water bodies. *Water Research*, 35(16): 3783–3790. https://doi.org/10.1016/S0043-1354(01)00112-9

**Köpke, U. & Nemecek, T.** 2010. Ecological services of faba bean. *Field Crops Research*, 115(3): 217–233. https://doi.org/10.1016/j.fcr.2009.10.012

**Koppelmäki, K., Helenius, J. & Schulte, R.P.** 2021. Nested circularity in food systems: A Nordic case study on connecting biomass, nutrient and energy flows from field scale to continent. *Resources, Conservation and Recycling*, 164: 105218. https://doi.org/10.1016/j.resconrec.2020.105218

**Koppelmäki, K., Parviainen, T., Virkkunen, E., Winquist, E., Schulte, R.P. & Helenius, J.** 2019. Ecological intensification by integrating biogas production into nutrient cycling: Modelling the case of agroecological symbiosis. *Agricultural Systems*, 170: 39–48. https://doi.org/10.1016/j.agsy.2018.12.007

**Koziol, L.., Lubin, T., & Bever, J. D.** 2024. An assessment of twenty-three mycorrhizal inoculants reveals limited viability of AM fungi, pathogen contamination, and negative microbial effect on crop growth for commercial products. *Applied Soil Ecology*, 202: 105559. https://doi.org/10.1016/j.apsoil.2024.105559

**Kuang, W., Gao, X., Tenuta, M. & Zeng, F.** 2021. A global meta-analysis of nitrous oxide emission from drip-irrigated cropping system. *Global Change Biology*, 27(14): 3244–3256. https://doi.org/10.1111/gcb.15636

**Kumar, A., Yang, T. & Sharma, M.P.** 2019. Greenhouse gas measurement from Chinese freshwater bodies: A review. *Journal of Cleaner Production*, 233: 368–378. https://doi.org/10.1016/j.jclepro.2019.06.052

**Kumar, N., Chaudhary, A., Ahlawat, O.P., Naorem, A., Upadhyay, G., Chhokar, R.S., Gill, S.C. *et al.*** 2023. Crop residue management challenges, opportunities and way forward for sustainable food-energy security in India: A review. *Soil and Tillage Research*, 228: 105641. https://doi.org/10.1016/j.still.2023.105641

**Kupper, T., Häni, C., Neftel, A., Kincaid, C., Bühler, M., Amon, B. & VanderZaag, A.** 2020. Ammonia and greenhouse gas emissions from slurry storage – A review. *Agriculture, Ecosystems & Environment*, 300: 106963. https://doi.org/10.1016/j.agee.2020.106963

**Ladha, J.K., Pathak, H., J. Krupnik, T., Six, J. & van Kessel, C.** 2005. Efficiency of fertilizer nitrogen in cereal production: Retrospects and prospects. In: *Advances in Agronomy*. pp. 85–156. Vol. 87. Academic Press. https://doi.org/10.1016/S0065-2113(05)87003-8

**Lal, R., Smith, P., Jungkunst, H.F., Mitsch, W.J., Lehmann, J., Nair, P.K.R., McBratney, A.B. *et al.*** 2018. The carbon sequestration potential of terrestrial ecosystems. *Journal of Soil and Water Conservation*, 73(6): 145A-152A. https://doi.org/10.2489/jswc.73.6.145A

**Landbrugsstyrelsen**. 2019. Vejledning om gødsknings og harmoniregler. Planperioden 1. august 2019 til 31. juli 2020. Landbrugsstyrelsen. https://lbst.dk/Media/638530103366163515/Vejledning_om_goedsknings-_og_harmoniregler_2018_2019_1version.pdf

**Lang, C.** 2022. Bioeconomy-from the Cologne paper to concepts for a global strategy. *EFB Bioeconomy Journal*, 2: 100038. https://doi.org/10.1016/j.bioeco.2022.100038

**Lapidus, D., Salem, M.E., Beach, R.H., Zayed, S. & Ortiz-Monasterio, I.** 2022. Greenhouse gas mitigation benefits and profitability of the GreenSeeker Handheld NDVI sensor: Evidence from Mexico. *Precision Agriculture*, 23(6): 2388–2406. https://doi.org/10.1007/s11119-022-09925-z

**Lasek, J.A. & Lajnert, R.** 2022. On the Issues of $NO_x$ as greenhouse gases: An ongoing discussion. *Applied Sciences*, 12(20): 10429. https://doi.org/10.3390/app122010429

**Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J. & Garnier, J.** 2014a. 50 year trends in nitrogen use efficiency of world cropping systems: The relationship between yield and nitrogen input to cropland. *Environmental Research Letters*, 9(10): 105011. http://dx.doi.org/10.1088/1748-9326/9/10/105011

**Lassaletta, L., Billen, G., Grizzetti, B., Garnier, J., Leach, A.M. & Galloway, J.N.** 2014b. Food and feed trade as a driver in the global nitrogen cycle: Fifty-year trends. *Biogeochemistry*, 118: 225–241. http://dx.doi.org/10.1007/s10533-013-9923-4

**Lassaletta, L., Billen, G., Garnier, J., Bouwman, L., Velazquez, E., Mueller, N.D. & Gerber, J.S.** 2016. Nitrogen use in the global food system: Past trends and future trajectories of agronomic performance, pollution, trade, and dietary demand. *Environmental Research Letters*, 11(9): 095007. https://doi.org/10.1088/1748-9326/11/9/095007

**Lassaletta, L., Sanz-Cobena, A., Aguilera, E., Quemada, M., Billen, G., Bondeau, A., Cayuela, M.L. *et al.*** 2021. Nitrogen dynamics in cropping systems under Mediterranean climate: A systemic analysis. *Environmental Research Letters*, 16(7): 073002. https://doi.org/10.1088/1748-9326/ac002c

**Lassaletta, L., Sanz-Cobeña, A., Pinsard, C., Ma, L., Spiegal, S. & Reidsma, P.** 2024. Special issue opening editorial: Reducing nitrogen waste through crop and livestock reconnection. *Agricultural Systems*, 214: 103816. https://doi.org/10.1016/j.agsy.2023.103816

**Lassaletta, L., Garnier, J., Quemada, M., Sanz-Cobeña, A., Mateo, A., Billen, G.** (forthcoming). Considering the whole rotation when estimating NUE indicators. In: L. Lassaletta & A. Sanz-Cobeña, eds. *Guidance Document on nitrogen use efficiency indicators across multiple scales*. INMS Guidance Document Series. Edinburgh, UK Centre for Ecology & Hydrology.

**Laura, F., Tamara, A., Müller, A., Hiroshan, H., Christina, D. & Serena, C.** 2020. Selecting sustainable sewage sludge reuse options through a systematic assessment framework: Methodology and case study in Latin America. *Journal of Cleaner Production*, 242: 118389. https://doi.org/10.1016/j.jclepro.2019.118389

**Lavado, R.S., Rodríguez, M.B. & Taboada, M.A.** 2005. Treatment with biosolids affects soil availability and plant uptake of potentially toxic elements. *Agriculture, Ecosystems & Environment*, 109(3): 360–364. https://doi.org/10.1016/j.agee.2005.03.010

**Lavallais, C.M. & Dunn, J.B.** 2023. Developing product level indicators to advance the nitrogen circular economy. *Resources, Conservation and Recycling*, 198: 107167. https://doi.org/10.1016/j.resconrec.2023.107167

**Le Dinh, P., van der Peet-Schwering, C.M., Ogink, N.W. & Aarnink, A.J.** 2022. Effect of diet composition on excreta composition and ammonia emissions from growing-finishing pigs. *Animals*, 12(3): 229. https://doi.org/10.3390/ani12030229

**Le Moal, M., Gascuel-Odoux, C., Ménesguen, A., Souchon, Y., Étrillard, C., Levain, A., Moatar, F. *et al.*** 2019. Eutrophication: A new wine in an old bottle? *Science of the Total Environment*, 651: 1–11. https://doi.org/10.1016/j.scitotenv.2018.09.139

**Le Noë, J., Billen, G., Lassaletta, L., Silvestre, M. & Garnier, J.** 2016. *La place du transport de denrées agricoles dans le cycle biogéochimique de l'azote en France: Un aspect de la spécialisation des territoires*. Cah. Agric., 25(1). https://doi.org/10.1051/cagri/2016002

**Leip, A., Bodirsky, B.L. & Kugelberg, S.** 2021. The role of nitrogen in achieving sustainable food systems for healthy diets. *Global Food Security*, 28: 100408. https://doi.org/10.1016/j.gfs.2020.100408

**Leip, A., Billen, G., Garnier, J., Grizzetti, B., Lassaletta, L., Reis, S., Simpson, D. *et al.*** 2015. Impacts of European livestock production: Nitrogen, sulphur, phosphorus and greenhouse gas emissions, land-use, water eutrophication and biodiversity. *Environmental Research Letters*, 10(11): 115004. https://doi.org/10.1088/1748-9326/10/11/115004

**Leip, A., Ledgard, S., Uwizeye, A., Palhares, J.C.P., Aller, M.F., Amon, B., Binder, M. *et al.*** 2019. The value of manure – Manure as co-product in life cycle assessment. *Journal of Environmental Management*, 241: 293–304. https://doi.org/10.1016/j.jenvman.2019.03.059

**Leip, A., Caldeira, C., Corrado, S., Hutchings, N.J., Lesschen, J.P., Schaap, M., de Vries, W., Westhoek, H. & van Grinsven, H.J.** 2022. Halving nitrogen waste in the European Union food systems requires both dietary shifts and farm level actions. *Global Food Security*, 35: 100648. https://doi.org/10.1016/j.gfs.2022.100648

**Leip, A., Wollgast, J., Kugelberg, S., Costa Leite, J., Maas, R.J., Mason, K.E. & Sutton, M.A.** (eds.). 2023. *Appetite for change: Food system options for nitrogen, environment and health.* Second European Nitrogen Assessment special report on nitrogen and food. Edinburgh, UK Centre for Ecology & Hydrology. https://doi.org/10.5281/zenodo.10406450

**Li, Y., Zhang, W., Ma, L., Huang, G., Oenema, O., Zhang, F. & Dou, Z.** 2013. An analysis of China's fertilizer policies: impacts on the industry, food security, and the environment. *Journal of Environmental Quality*, 42(4): 972–981. https://doi.org/10.2134/jeq2012.0465

**Li, S., Liu, Y., Wang, J., Yang, L., Zhang, S., Xu, C. & Ding, W.** 2017. Soil acidification aggravates the occurrence of bacterial wilt in South China. *Frontiers in Microbiology*, 8: 703. https://doi.org/10.3389/fmicb.2017.00703

**Li, P., Li, Y., Xu, L., Zhang, H., Shen, X., Xu, H., Jiao, J., Li, H. & Hu, F.** 2021. Crop yield-soil quality balance in double cropping in China's upland by organic amendments: A meta-analysis. *Geoderma*, 403: 115197. https://doi.org/10.1016/j.geoderma.2021.115197

**Liang, H., Zhang, X., Han, J., Liao, Y., Liu, Y. & Wen, X.** 2019. Integrated N management improves nitrogen use efficiency and economics in a winter wheat-summer maize multiple-cropping system. *Nutrient Cycling in Agroecosystems*, 115(3): 313–329. https://doi.org/10.1007/s10705-019-10014-3

**Lin, B., Xu, M. & Wang, X.** 2022. Mitigation of greenhouse gas emissions in China's agricultural sector: Current status and future perspectives. *Chinese Journal of Eco-Agriculture*, 30(4): 500–515. https://dx.doi.org/10.12357/cjea.20210843

**Lin, J., Compton, J.E., Clark, C., Bittman, S., Schwede, D., Homann, P.S., Kiffney, P. *et al.*** 2020. Key components and contrasts in the nitrogen budget across a US-Canadian transboundary watershed. *Journal of Geophysical Research: Biogeosciences*, 125(9): e2019JG005577. http://dx.doi.org/10.1029/2019JG005577

**Liu, L. & Greaver, T.L.** 2009. A review of nitrogen enrichment effects on three biogenic GHGs: the $CO_2$ sink may be largely offset by stimulated $N_2O$ and $CH_4$ emissions. *Ecology Letters*, 12(10): 1103–1117. https://doi.org/10.1111/j.1461-0248.2009.01351.x

**Liu, L. & Greaver, T.L.** 2010. A global perspective on belowground carbon dynamics under nitrogen enrichment. *Ecology Letters*, 13(7): 819–828. https://doi.org/10.1111/j.1461-0248.2010.01482.x

**Liu, C., Wang, K., Meng, S., Zheng, X., Zhou, Z., Han, S., Chen, D. & Yang, Z.** 2011. Effects of irrigation, fertilization and crop straw management on nitrous oxide and nitric oxide emissions from a wheat-maize rotation field in northern China. *Agriculture, Ecosystems & Environment*, 140(1): 226–233. https://doi.org/10.1016/j.agee.2010.12.009

**Liu, S., Yang, J.Y., Zhang, X.Y., Drury, C.F., Reynolds, W.D. & Hoogenboom, G.** 2013. Modelling crop yield, soil water content and soil temperature for a soybean-maize rotation under conventional and conservation tillage systems in north-east China. *Agricultural Water Management*, 123: 32–44. https://doi.org/10.1016/j.agwat.2013.03.001

**Liu, F., Zhang, Q., Ronald J. van der A., Zheng, B., Tong, D., Yan, L., Zheng, Y. & He, K.** 2016. Recent reduction in $NO_x$ emissions over China: synthesis of satellite observations and emission inventories. *Environmental Research Letters*, 11(11): 114002. https://doi.org/10.1088/1748-9326/11/11/114002

**Liu, Q., Zhang, Y., Liu, B., Amonette, J.E., Lin, Z., Liu, G., Ambus, P. & Xie, Z.** 2018. How does biochar influence soil N cycle? A meta-analysis. *Plant and Soil*, 426(1): 211–225. https://doi.org/10.1007/s11104-018-3619-4

**Liu, B., Wang, X., Ma, L., Chadwick, D. & Chen, X.** 2021. Combined applications of organic and synthetic nitrogen fertilizers for improving crop yield and reducing reactive nitrogen losses from China's vegetable systems: A meta-analysis. *Environmental Pollution*, 269: 116143. https://doi.org/10.1016/j.envpol.2020.116143

**Lobell, D.B., Ortiz-Monasterio, J.I. & Asner, G.P.** 2004. Relative importance of soil and climate variability for nitrogen management in irrigated wheat. *Field Crops Research*, 87(2–3): 155–165. https://doi.org/10.1016/j.fcr.2003.10.004

**López-Bellido, R.J. & López-Bellido, L.** 2001. Efficiency of nitrogen in wheat under Mediterranean conditions: effect of tillage, crop rotation and N fertilization. *Field Crops Research*, 71(1): 31–46. https://doi.org/10.1016/S0378-4290(01)00146-0

**Lorenz, K. & Lal, R.** 2018. *Carbon Sequestration in agricultural ecosystems*. Cham, Springer International Publishing. https://doi.org/10.1007/978-3-319-92318-5

**Löw, P., Karatay, Y.N. & Osterburg, B.** 2020. Nitrogen use efficiency on dairy farms with different grazing systems in northwestern Germany. *Environmental Research Communications*, 2(10): 105002. https://doi.org/10.1088/2515-7620/abc098

**Lu, X., Vitousek, P.M., Mao, Q., Gilliam, F.S., Luo, Y., Turner, B.L., Zhou, G. & Mo, J.** 2021. Nitrogen deposition accelerates soil carbon sequestration in tropical forests. *Proceedings of the National Academy of Sciences*, 118(16): e2020790118. https://doi.org/10.1073/pnas.2020790118

**Lu, X., Che, Y., Rejesus, R.M., Goodwin, B.K., Ghosh, S.K. & Paudel, J.** 2023. Unintended environmental benefits of crop insurance: Nitrogen and phosphorus in water bodies. *Ecological Economics*, 204: 107657. https://doi.org/10.1016/j.ecolecon.2022.107657

**Ludemann, C.I., Wanner, N., Chivenge, P., Dobermann, A., Einarsson, R., Grassini, P., Gruere, A. *et al.*** 2024. A global FAOSTAT reference database of cropland nutrient budgets and nutrient use efficiency (1961–2020): Nitrogen, phosphorus and potassium. *Earth System Science Data*, 16(1): 525–541. https://doi.org/10.5194/essd-16-525-2024

**Lugato, E., Leip, A. & Jones, A.** 2018. Mitigation potential of soil carbon management overestimated by neglecting $N_2O$ emissions. *Nature Climate Change*, 8(3): 219–223. https://doi.org/10.1038/s41558-018-0087-z

**Luo, J. & Ledgard, S.** 2021. New Zealand dairy farm systems and key environmental effects. *Frontiers of Agricultural Science and Engineering*, 8(1): 148. https://doi.org/10.15302/J-FASE-2020372

**Machiwal, D., Jha, M.K., Singh, V.P. & Mohan, C.** 2018. Assessment and mapping of groundwater vulnerability to pollution: Current status and challenges. *Earth-Science Reviews*, 185: 901–927. https://doi.org/10.1016/j.earscirev.2018.08.009

**Mak, T.M.W., Xiong, X., Tsang, D.C.W., Yu, I.K.M. & Poon, C.S.** 2020. Sustainable food waste management towards circular bioeconomy: Policy review, limitations and opportunities. *Bioresource Technology*, 297: 122497. https://doi.org/10.1016/j.biortech.2019.122497

**Malomo, G.A., Bolu, S.A., Madugu, A.S. & Usman, Z.S.** 2018. Nitrogen emissions and mitigation strategies in chicken production. *Animal Husbandry and Nutrition*, 43: 43–62. http://dx.doi.org/10.5772/intechopen.74966

**Maredia, M.K., Reyes, B. & DeYoung, D.** 2014. *Farmer perspective on the use of and demand for seeds of improved bean varieties: Results of beneficiary surveys in Guatemala, Honduras and Nicaragua*. Staff Paper Series 196540, Michigan State University, Department of Agricultural, Food, and Resource Economics. http://doi.org/10.22004/ag.econ.196540

**Marouli, C.** 2024. Food waste interventions: Barriers on the way to sustainable food systems. *Sustainable Development*, 32(3). https://doi.org/10.1002/sd.2660

**Marschner, H.** 1995. *Mineral Nutrition of Higher Plants. Second Edition.* Academic Press, Elsevier Ltd. https://doi.org/10.1016/C2009-0-02402-7

**Martin, T.M.P., Esculier, F., Levavasseur, F. & Houot, S.** 2022. Human urine-based fertilizers: A review. *Critical Reviews in Environmental Science and Technology*, 52(6): 890–936. https://doi.org/10.1080/10643389.2020.1838214

**Martinez-Feria, R.A., Castellano, M.J., Dietzel, R.N., Helmers, M.J., Liebman, M., Huber, I. & Archontoulis, S.V.** 2018. Linking crop- and soil-based approaches to evaluate system nitrogen-use efficiency and tradeoffs. *Agriculture, Ecosystems & Environment*, 256: 131–143. https://doi.org/10.1016/j.agee.2018.01.002

**Martin-Hurtado, R. & Nolasco, D.** 2017. *Managing Wastewater as a Resource in Latin America and the Caribbean. Towards a Circular Economy Approach.* World Water Week, Stockholm International Water Institute. https://programme.worldwaterweek.org/Content/ProposalResources/allfile/managing_wastewater_as_a_resource_in_lac.pdf

**Maskell, L.C., Smart, S.M., Bullock, J.M., Thompson, K. & Stevens, C.J.** 2010. Nitrogen deposition causes widespread loss of species richness in British habitats. *Global Change Biology*, 16(2): 671–679. https://doi.org/10.1111/j.1365-2486.2009.02022.x

**Matson, P.A., Naylor, R. & Ortiz-Monasterio, I.** 1998. Integration of environmental, agronomic and economic aspects of fertilizer management. *Science*, 280(5360): 112–115. https://doi.org/10.1126/science.280.5360.112

**Maúre, E.D.R., Terauchi, G., Ishizaka, J., Clinton, N. & DeWitt, M.** 2021. Globally consistent assessment of coastal eutrophication. *Nature Communications*, 12(1): 6142. https://doi.org/10.1038/s41467-021-26391-9

**McBride, M., Loyola, C., Papadimitriou, C. & Patterson, P.** 2021. *No food left behind-benefits and trade-offs of food waste-to-feed pathways*. Washington, DC, World Wildlife Fund. https://files.worldwildlife.org/wwfcmsprod/files/Publication/file/2q8g6qmx4s_WWF_NoFoodIV_Waste_to_Feed_Pathways.pdf

**McLellan, E.L., Cassman, K.G., Eagle, A.J., Woodbury, P.B., Sela, S., Tonitto, C., Marjerison, R.D. & Harold, M. van Es.** 2018. The nitrogen balancing act: Tracking the environmental performance of food production. *BioScience*, 68(3): 194–203. https://doi.org/10.1093/biosci/bix164

**McSherry, M.E. & Ritchie, M.E.** 2013. Effects of grazing on grassland soil carbon: A global review. *Global Change Biology*, 19(5): 1347–1357. https://doi.org/10.1111/gcb.12144

**Méité, R., Artner-Nehls, A. & Uthes, S.** 2024. Farm adaptation to stricter nutrient management legislation and the implications for future livestock production: A review. *Nutrient Cycling in Agroecosystems*. https://doi.org/10.1007/s10705-024-10341-0

**Mekonnen, M.M., Neale, C.M.U., Ray, C., Erickson, G.E. & Hoekstra, A.Y.** 2019. Water productivity in meat and milk production in the US from 1960 to 2016. *Environment International*, 132: 105084. https://doi.org/10.1016/j.envint.2019.105084

**Menegat, S., Ledo, A. & Tirado, R.** 2022. Greenhouse gas emissions from global production and use of nitrogen synthetic fertiliszers in agriculture. *Scientific Reports*, 12(1): 14490. https://doi.org/10.1038/s41598-022-18773-w

**Messina, J.P., Peter, B.G. & Snapp, S.S.** 2017. Re-evaluating the Malawian farm input subsidy programme. *Nature Plants*, 3(4): 17013. https://doi.org/10.1038/nplants.2017.13

**Millar, N., Urrea, A., Kahmark, K., Shcherbak, I., Robertson, G.P. & Ortiz-Monasterio, I.** 2018. Nitrous oxide ($N_2O$) flux responds exponentially to nitrogen fertilizer in irrigated wheat in the Yaqui Valley, Mexico. *Agriculture, Ecosystems & Environment*, 261: 125–132. https://doi.org/10.1016/j.agee.2018.04.003

**Misstear, B., Banks, D. & Clark, L.** 2017. *Water Wells and Boreholes*. First edition. Wiley. https://doi.org/10.1002/9781119080176

**MOA**. 2019. *China Agriculture Yearbook 2018*. https://www.chinayearbooks.com/2020/07

**Møller, H.B., Sørensen, P., Olesen, J.E., Petersen, S.O., Nyord, T. & Sommer, S.G.** 2022. Agricultural Biogas Production – Climate and Environmental Impacts. *Sustainability*, 14(3). https://doi.org/10.3390/su14031849

**Moraine, M., Duru, M. & Therond, O.** 2017. A social-ecological framework for analyzing and designing integrated crop-livestock systems from farm to territory levels. *Renewable Agriculture and Food Systems*, 32(1): 43–56. https://dx.doi.org/10.1017/S1742170515000526

**Mottet, A., de Haan, C., Falcucci, A., Tempio, G., Opio, C. & Gerber, P.** 2017. Livestock: On our plates or eating at our table? A new analysis of the feed/food debate. *Global Food Security*, 14: 1–8. https://doi.org/10.1016/j.gfs.2017.01.001

**Mourouzidou, S., Ntinas, G.K., Tsaballa, A. & Monokrousos, N.** 2023. Introducing the power of plant growth promoting microorganisms in soilless systems: A promising alternative for sustainable agriculture. *Sustainability*, 15(7). https://doi.org/10.3390/su15075959

**Mueller, N.D., Gerber, J.S., Johnston, M., Ray, D.K., Ramankutty, N. & Foley, J.A.** 2012. Closing yield gaps through nutrient and water management. *Nature*, 490(7419): 254–257. https://doi.org/10.1038/nature11420

**Mueller, N.D., Lassaletta, L., Runck, B.C., Billen, G., Garnier, J. & Gerber, J.S.** 2017. Declining spatial efficiency of global cropland nitrogen allocation. *Global Biogeochemical Cycles*, 31(2): 245–257. https://doi.org/10.1002/2016GB005515

**Muscat, A., de Olde, E.M., Ripoll-Bosch, R., Van Zanten, H.H.E., Metze, T.A.P., Termeer, C.J.A.M., van Ittersum, M.K. & de Boer, I.J.M.** 2021. Principles, drivers and opportunities of a circular bioeconomy. *Nature Food*, 2(8): 561–566. https://doi.org/10.1038/s43016-021-00340-7

**Musigwa, S., Morgan, N., Swick, R.A., Cozannet, P. & Wu, S.-B.** 2020. Energy dynamics, nitrogen balance, and performance in broilers fed high- and reduced-CP diets. *Journal of Applied Poultry Research*, 29(4): 830–841. https://doi.org/10.1016/j.japr.2020.08.001

**Mutiro, K. & Murwira, H.** 2004. The profitability of manure use on maize in the smallholder sector of Zimbabwe. In: A. Bationo, ed. *Managing nutrient cycles to sustain soil fertility in sub-Saharan Africa.* Nairobi, Academy Science Publishers (ASP); Centro Internacional de Agricultura Tropical (CIAT). https://hdl.handle.net/10568/55353

**Nadtochii, L., Orazov, A., Muradova, M., Bozymov, K., Japarova, A. & Baranenko, D.** 2018. Comparison of the energy efficiency of production of camel's and cow's milk resources. *Energy Procedia*, 147: 510–517. https://doi.org/10.1016/j.egypro.2018.07.064

**Nasielski, J., Grant, B., Smith, W., Niemeyer, C., Janovicek, K. & Deen, B.** 2020. Effect of nitrogen source, placement and timing on the environmental performance of economically optimum nitrogen rates in maize. *Field Crops Research*, 246: 107686. https://doi.org/10.1016/j.fcr.2019.107686

**Nath, P., Ojha, A., Debnath, S., Sharma, M., Nayak, P.K., Sridhar Kandi & Inbaraj, B.** 2023. Valorization of food waste as animal feed: A step towards sustainable food waste management and circular bioeconomy animal feed *Animals*, 13: 1366. https://doi.org/10.3390/ani13081366

**Nazari-Sharabian, M., Ahmad, S. & Karakouzian, M.** 2018. Climate change and eutrophication: A short review. *Engineering, Technology and Applied Science Research*, 8(6): 3668–3672. https://doi.org/10.48084/etasr.2392

**Ndambi, O.A., Pelster, D.E., Owino, J.O., De Buisonje, F. & Vellinga, T.** 2019. Manure management practices and policies in sub-Saharan Africa: Implications on manure quality as a fertilizer. *Frontiers in Sustainable Food Systems*, 3: 29. https://doi.org/10.3389/fsufs.2019.00029

**Nemecek, T., von Richthofen, J.-S., Dubois, G., Casta, P., Charles, R. & Pahl, H.** 2008. Environmental impacts of introducing grain legumes into European crop rotations. *European Journal of Agronomy*, 28(3): 380–393. https://doi.org/10.1016/j.eja.2007.11.004

**Nepal, M., Ashfaq, M., Sharma, B.R., Shrestha, M.S., Khadgi, V.R. & Bruno Soares, M.** 2024. Impact of weather and climate advisories on agricultural outcomes in Pakistan. *Scientific Reports*, 14(1): 1036. https://doi.org/10.1038/s41598-023-51066-4

**Ngatunga, E.L., Dondeyne, S., Cools, N., Dondeyne, S., Deckers, J.A. & Merckx, R.** 2001. Buffering capacity of cashew soils in South Eastern Tanzania. *Soil Use and Management*, 17(3): 155–162. https://doi.org/10.1111/j.1475-2743.2001.tb00022.x

**Nhlengethwa, S., Thangata, P., Muthini, D., Djido, A., Njiwa, D. & Nwafor, A.** 2023. *Review of agricultural subsidy programmes in Sub-Saharan Africa: The impact of the Russia-Ukraine war*. AGRA Hub for Agricultural Policy Action, Policy Brief 3. https://agra.org/wp-content/uploads/2023/01/HAPA-Review-of-Agricultural-Subsidy-Programmes-in-Sub-Saharan-Africa.pdf

**Nilsson, J.** 1988. Critical Loads for Sulphur and Nitrogen. In: P. Mathy, ed. *Air Pollution and Ecosystems.* Dordrecht, Springer. https://doi.org/10.1007/978-94-009-4003-1_11

**Nin, A., Freiría, H. & Muñoz, G.** 2019. *Productivity and efficiency in grassland-based livestock production in Latin America: The cases of Uruguay and Paraguay*. IDB Working Paper Series No. IDB-WP-1024. Washington, Inter-American Development Bank. https://doi.org/10.18235/0001924%0A

**Niu, S., Classen, A.T., Dukes, J.S., Kardol, P., Liu, L., Luo, Y., Rustad, L. *et al.*** 2016. Global patterns and substrate-based mechanisms of the terrestrial nitrogen cycle. *Ecology Letters*, 19(6): 697–709. https://doi.org/10.1111/ele.12591

**Nixon, S.W.** 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia*, 41(1): 199–219. https://doi.org/10.1080/00785236.1995.10422044

**Nowroz, F., Hasanuzzaman, M., Siddika, A., Parvin, K., Caparros, P.G., Nahar, K. & Prasad, P.V.V.** 2024. Elevated tropospheric ozone and crop production: Potential negative effects and plant defense mechanisms. *Frontiers in Plant Science*, 14. https://www.frontiersin.org/journals/plant-science/articles/10.3389/fpls.2023.1244515

**NSIR (National Institute of Statistics of Rwanda)**. 2020. Agriculture household survey data 2020. Kigali, National Institute Of Statistics Of Rwanda. https://microdata.statistics.gov.rw/index.php/catalog/101

**Nwankwegu, A.S., Li, Y., Huang, Y., Wei, J., Norgbey, E., Sarpong, L., Lai, Q. & Wang, K.** 2019. Harmful algal blooms under changing climate and constantly increasing anthropogenic actions: The review of management implications. *3 Biotech*, 9(12): 449. https://doi.org/10.1007/s13205-019-1976-1

**Nziguheba, G., van Heerwaarden, J. & Vanlauwe, B.** 2021. Quantifying the prevalence of (non)-response to fertilizers in sub-Saharan Africa using on-farm trial data. *Nutrient Cycling in Agroecosystems*, 121: 257–269. https://doi.org/10.1007/s10705-021-10174-1

**O'Dea, C.B., Anderson, S., Sullivan, T., Landers, D. & Casey, C.F.** 2017. Impacts to ecosystem services from aquatic acidification: using FEGS-CS to understand the impacts of air pollution. *Ecosphere*, 8(5): e01807. https://doi.org/10.1002/ecs2.1807

**O'Donovan, M., Hennessy, D. & Creighton, P.** 2021. Ruminant grassland production systems in Ireland. *Irish Journal of Agricultural and Food Research*, 59(2). https://doi.org/10.15212/ijafr-2020-0118

**Oenema, O.** 2004. Governmental policies and measures regulating nitrogen and phosphorus from animal manure in European Agriculture. *Journal of Animal Science*, 82 E-Suppl: E196-206. https://doi.org/10.2527/2004.8213_supplE196x

**Oenema, O.** 2006. Nitrogen budgets and losses in livestock systems. *International Congress Series*, 1293: 262–271. https://doi.org/10.1016/j.ics.2006.02.040

**Oenema, O., Bleeker, A., Braathen, N.A., Budňakova, M., Bull, K., Čermak, P., Geupel, M. *et al.*** 2011. Nitrogen in current European policies. In: M. Sutton, C.M. Howard, J.W. Erisman, G. Billen, A. Bleeker, P. Grennfelt, H. Grinsven & B. Grizzetti, eds. *The European Nitrogen Assessment*. Cambridge University Press. https://www.cambridge.org/core/books/abs/european-nitrogen-assessment/nitrogen-in-current-european-policies/D9816BF2A75443D6F332C32D22CFD350

**Oenema, O., Bannink, A., Sommer, S.G., van Groenigen, J. & Velthof, G.** 2008. Gaseous nitrogen emissions from livestock farming systems. In: J.L. Hatfield & R.F. Follett, eds. *Nitrogen in the environment sources, problems, and management*. London, Elsevier. https://doi.org/10.1016/B978-0-12-374347-3.00012-3

**Oita, A., Malik, A., Kanemoto, K., Geschke, A., Nishijima, S. & Lenzen, M.** 2016. Substantial nitrogen pollution embedded in international trade. *Nature Geoscience*, 9(2): 111–115. https://doi.org/10.1038/ngeo2635

**Omara, P., Aula, L., Oyebiyi, F. & Raun, W.R.** 2019. World cereal nitrogen use efficiency trends: review and current knowledge. *Agrosystems, Geosciences & Environment*, 2(1): 1–8. https://doi.org/10.2134/age2018.10.0045

**Oosting, S., van der Lee, J., Verdegem, M., de Vries, M., Vernooij, A., Bonilla-Cedrez, C. & Kabir, K.** 2022. Farmed animal production in tropical circular food systems. *Food Security*, 14(1): 273–292. https://doi.org/10.1007/s12571-021-01205-4

**Ormerod, S.J. & Durance, I.** 2009. Restoration and recovery from acidification in upland Welsh streams over 25 years. *Journal of Applied Ecology*, 46(1): 164–174. https://doi.org/10.1111/j.1365-2664.2008.01587.x

**Ortiz-Monasterio, J.I. & Raun, W.** 2007. Reduced nitrogen and improved farm income for irrigated spring wheat in the Yaqui Valley, Mexico, using sensor-based nitrogen management. *The Journal of Agricultural Science*, 145(3): 215–222. https://doi.org/10.1017/S0021859607006995

**Ortiz-Monasterio, J.I., Sayre, K.D., Rajaram, S. & McMahon, M.** 1997. Genetic progress in wheat yield and nitrogen use efficiency under four nitrogen rates. *Crop Science*, 37(3):cropsci1997.0011183X003700030033x. https://doi.org/10.2135/cropsci1997.0011183X003700030033x

**Otte, J., Pica-Ciamarra, U. & Morzaria, S.** 2019. A Comparative Overview of the Livestock-Environment Interactions in Asia and Sub-saharan Africa. *Frontiers in Veterinary Science*, 6. https://www.frontiersin.org/articles/10.3389/fvets.2019.00037

**Pahalvi, H.N., Rafiya, L., Rashid, S., Nisar, B. & Kamili, A.N.** 2021. Chemical fertilizers and their impact on soil health. In: G.H. Dar, R.A. Bhat, M.A. Mehmood & K.R. Hakeem, eds. *Microbiota and Biofertilizers, Vol 2: Ecofriendly Tools for Reclamation of Degraded Soil Environs*. pp. 1–20. Cham, Springer International Publishing. https://doi.org/10.1007/978-3-030-61010-4_1

**Pan, B., Lam, S.K., Mosier, A., Luo, Y. & Chen, D.** 2016. Ammonia volatilization from synthetic fertilizers and its mitigation strategies: A global synthesis. *Agriculture, Ecosystems & Environment*, 232: 283–289. https://doi.org/10.1016/j.agee.2016.08.019

**Paramesh, V., Sreekanth, G.B., Chakurkar, Eaknath.B., Chethan Kumar, H.B., Gokuldas, P., Manohara, K.K., Ramdas Mahajan, G. *et al.*** 2020. Ecosystem network analysis in a smallholder integrated crop–livestock system for coastal lowland situation in tropical humid conditions of India. *Sustainability*, 12(12). https://doi.org/10.3390/su12125017

**Parfitt, J., Barthel, M. & Macnaughton, S.** 2010. Food waste within food supply chains: Quantification and potential for change to 2050. *Philosophical Transactions of the Royal Society B: Biological sciences*, 365(1554): 3065–3081. https://doi.org/10.1098/rstb.2010.0126

**Parton, W.J., Ojima, D.S., Cole, C.V. & Schimel, D.S.** 2015. A general model for soil organic matter dynamics: Sensitivity to litter chemistry, texture and management. In: R.B. Bryant & R.W. Arnold, eds. *SSSA Special Publications*. pp. 147–167. Madison, WI, USA, Soil Science Society of America. https://doi.org/10.2136/sssaspecpub39.c9

**Patel, A., Mungray, A.A. & Mungray, A.K.** 2020. Technologies for the recovery of nutrients, water and energy from human urine: A review. *Chemosphere*, 259: 127372. https://doi.org/10.1016/j.chemosphere.2020.127372

**Paul, B.K., Butterbach-Bahl, K., Notenbaert, A., Nduah Nderi, A. & Ericksen, P.** 2021. Sustainable livestock development in low- and middle-income countries: Shedding light on evidence-based solutions. *Environmental Research Letters*, 16(1): 011001. https://doi.org/10.1088/1748-9326/abc278

**Pedersen, M.F., Gyldengren, J.G., Pedersen, S.M., Diamantopoulos, E., Gislum, R. & Styczen, M.E.** 2021. A simulation of variable rate nitrogen application in winter wheat with soil and sensor information – An economic feasibility study. *Agricultural Systems*, 192: 103147. https://doi.org/10.1016/j.agsy.2021.103147

**Peng, Y., Li, F., Zhou, G., Fang, K., Zhang, D., Li, C., Yang, G. *et al.*** 2017. Linkages of plant stoichiometry to ecosystem production and carbon fluxes with increasing nitrogen inputs in an alpine steppe. *Global Change Biology*, 23(12): 5249–5259. https://doi.org/10.1111/gcb.13789

**Peng, Y., Peng, Z., Zeng, X. & Houx, J.H.** 2019. Effects of nitrogen-phosphorus imbalance on plant biomass production: A global perspective. *Plant and Soil*, 436(1): 245–252. https://doi.org/10.1007/s11104-018-03927-5

**Peoples, M., Boddey, R. & Herridge, D.** 2002. Quantification of Nitrogen Fixation. pp. 357–389. https://doi.org/10.1016/B978-044450965-9/50013-6

**Perming, E.** 2012. Nitrogen Footprint Vs. Life Cycle Impact Assessment methods – A comparison of the methods in a case study. Sweden, Lund University. MSc thesis.

**Peters, C.J., Picardy, J.A., Darrouzet-Nardi, A. & Griffin, T.S.** 2014. Feed conversions, ration compositions and land use efficiencies of major livestock products in U.S. agricultural systems. *Agricultural Systems*, 130: 35–43. https://doi.org/10.1016/j.agsy.2014.06.005

**Petersen, S.O., Regina, K., Pöllinger, A., Rigler, E., Valli, L., Yamulki, S., Esala, M. *et al.*** 2006. Nitrous oxide emissions from organic and conventional crop rotations in five European countries. *Mitigation of Greenhouse Gas Emissions from Livestock Production*, 112(2): 200–206. https://doi.org/10.1016/j.agee.2005.08.021

**Peterson, C.B., El Mashad, H.M., Zhao, Y., Pan, Y. & Mitloehner, F.M.** 2020. Effects of SOP lagoon additive on gaseous emissions from stored liquid dairy manure. *Sustainability*, 12(4): 1393. https://doi.org/10.3390/su12041393

**Pinder, R.W., Davidson, E.A., Goodale, C.L., Greaver, T.L., Herrick, J.D. & Liu, L.** 2012. Climate change impacts of US reactive nitrogen. *Proceedings of the National Academy of Sciences*, 109(20): 7671–7675. https://doi.org/10.1073/pnas.1114243109

**Pinder, R.W., Bettez, N.D., Bonan, G.B., Greaver, T.L., Wieder, W.R., Schlesinger, W.H. & Davidson, E.A.** 2013. Impacts of human alteration of the nitrogen cycle in the US on radiative forcing. *Biogeochemistry*, 114(1): 25–40. https://doi.org/10.1007/s10533-012-9787-z

**Piñeiro, V., Arias, J., Dürr, J., Elverdin, P., Ibáñez, A.M., Kinengyere, A., Opazo, C.M. *et al.*** 2020. A scoping review on incentives for adoption of sustainable agricultural practices and their outcomes. *Nature Sustainability*, 3(10): 809–820. https://www.nature.com/articles/s41893-020-00617-y

**Pires, M.V., Da Cunha, D.A., De Matos Carlos, S. & Costa, M.H.** 2015. Nitrogen-use efficiency, nitrous oxide emissions and cereal production in Brazil: current trends and forecasts. *PloS ONE*, 10(8): e0135234. https://doi.org/10.1371/journal.pone.0135234

**Poore, J. & Nemecek, T.** 2018. Reducing food's environmental impacts through producers and consumers. *Science*, 360(6392): 987–992. https://doi.org/10.1126/science.aaq0216

**Prandi, B., Faccini, A., Lambertini, F., Bencivenni, M., Jorba, M., van Droogenbroek, B., Bruggeman, G. *et al.*** 2019. Food wastes from agrifood industry as possible sources of proteins: A detailed molecular view on the composition of the nitrogen fraction, amino acid profile and racemisation degree of 39 food waste streams. *Food Chemistry*, 286: 567–575. https://doi.org/10.1016/j.foodchem.2019.01.166

**Pyett, S., Jenkins, W., van Mierlo, B., Trindade, L.M., Welch, D. & van Zanten, H.H.E.** (eds.) 2023. *Our future proteins: A diversity of perspectives*. Amsterdam, VU University Press.

**Qiao, C., Liu, L., Hu, S., Compton, J.E., Greaver, T.L. & Li, Q.** 2015. How inhibiting nitrification affects nitrogen cycle and reduces environmental impacts of anthropogenic nitrogen input. *Global change biology*, 21(3): 1249–1257. https://doi.org/10.1111/gcb.12802

**Quan, Z., Zhang, X., Fang, Y. & Davidson, E.A.** 2021. Different quantification approaches for nitrogen use efficiency lead to divergent estimates with varying advantages. *Nature Food*, 2(4): 241–245. https://doi.org/10.1038/s43016-021-00263-3

**Quemada, M. & Lassaletta, L.** 2024. Fertilizer dependency: a new indicator for assessing the sustainability of agrosystems beyond nitrogen use efficiency. *Agronomy for Sustainable Development*, 44(5): 44. https://doi.org/10.1007/s13593-024-00978-0

**Quemada, M., Baranski, M., Nobel-De Lange, M.N.J., Vallejo, A. & Cooper, J.M.** 2013. Meta-analysis of strategies to control nitrate leaching in irrigated agricultural systems and their effects on crop yield. *Agriculture, Ecosystems & Environment*, 174: 1–10. https://doi.org/10.1016/j.agee.2013.04.018

**Quemada, M., Alonso-Ayuso, M., Castellano-Hinojosa, A., Bedmar, E.J., Gabriel, J.L., García González, I., Valentín, F. & Calvo, M.** 2019. Residual effect of synthetic nitrogen fertilizers and impact on Soil Nitrifiers. *European Journal of Agronomy*, 109: 125917. https://doi.org/10.1016/j.eja.2019.125917

**Quemada, M., Lassaletta, L., Jensen, L.S., Godinot, O., Brentrup, F., Buckley, C., Foray, S. *et al.*** 2020a. Exploring nitrogen indicators of farm performance among farm types across several European case studies. *Agricultural Systems*, 177: 102689. https://doi.org/10.1016/j.agsy.2019.102689

**Quemada, M., Lassaletta, L., Leip, A., Jones, A. & Lugato, E.** 2020b. Integrated management for sustainable cropping systems: Looking beyond the greenhouse balance at the field scale. *Global Change Biology*, 26(4): 2584–2598. https://doi.org/10.1111/gcb.14989

**Rahimi, J., Fillol, E., Mutua, J.Y., Cinardi, G., Robinson, T.P., Notenbaert, A.M.O., Ericksen, P.J., Graham, M.W. & Butterbach-Bahl, K.** 2022. A shift from cattle to camel and goat farming can sustain milk production with lower inputs and emissions in north sub-Saharan Africa's drylands. *Nature Food*, 3(7): 523–531. https://doi.org/10.1038/s43016-022-00543-6

**Rajeh, C., Saoud, I.P., Kharroubi, S., Naalbandian, S. & Abiad, M.G.** 2021. Food loss and food waste recovery as animal feed: A systematic review. *Journal of Material Cycles and Waste Management*, 23(1): 1–17. https://doi.org/10.1007/s10163-020-01102-6

**Ramankutty, N., Mehrabi, Z., Waha, K., Jarvis, L., Kremen, C., Herrero, M. & Rieseberg, L.H.** 2018. Trends in global agricultural land use: Implications for environmental health and food security. *Annual Review of Plant Biology*, 69: 789–815. https://doi.org/10.1146/annurev-arplant-042817-040256

**Ramirez-Corredores, M.M., Rollins, H.W., Morco, R.P., Zarzana, C.A. & Diaz, L.A.** 2023. Radiation-induced dry reforming: A negative emission process. *Journal of Cleaner Production*, 429: 139539. https://doi.org/10.1016/j.jclepro.2023.139539

**Raun, W.R. & Schepers, J.S.** 2008. Nitrogen management for improved use efficiency. In: *Nitrogen in Agricultural Systems*. pp. 675–693. John Wiley & Sons, Ltd. https://doi.org/10.2134/agronmonogr49.c17

**Reimer, A.P., Denny, R.C.H. & Stuart, D.** 2018. The impact of federal and State conservation programmes on farmer nitrogen management. *Environmental Management*, 62(4): 694–708. https://doi.org/10.1007/s00267-018-1083-9

**Reis, S., Bekunda, M., Howard, C.M., Karanja, N., Winiwarter, W., Yan, X., Bleeker, A. & Sutton, M.A.** 2016. Synthesis and review: Tackling the nitrogen management challenge: from global to local scales. *Environmental Research Letters*, 11(12): 120205. http://doi.org/10.1088/1748-9326/11/12/120205

**Ren, K., Xu, M., Li, R., Zheng, L., Liu, S., Reis, S., Wang, H. *et al.*** 2022. Optimizing nitrogen fertilizer use for more grain and less pollution. *Journal of Cleaner Production*, 360: 132180. https://doi.org/10.1016/j.jclepro.2022.132180

**Rengel, Z.** 2011. Soil pH, soil health and climate change. In: B.P. Singh, A.L. Cowie & K.Y. Chan, eds. *Soil Health and Climate Change*. pp. 69–85. Vol. 29. Soil Biology. Berlin, Heidelberg, Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-20256-8_4

**Renwick, L.L.R., Bowles, T.M., Deen, W. & Gaudin, A.C.M.** 2019. Potential of Increased temporal crop diversity to improve resource use efficiencies: Exploiting water and nitrogen linkages. In: G. Lemaire, P.C.D.F. Carvalho, S. Kronberg & S. Recous, eds. *Agroecosystem Diversity*. Academic Press. https://doi.org/10.1016/B978-0-12-811050-8.00004-2

**Repeta, D.J., Ferrón, S., Sosa, O.A., Johnson, C.G., Repeta, L.D., Acker, M., DeLong, E.F. & Karl, D.M.** 2016. Marine methane paradox explained by bacterial degradation of dissolved organic matter. *Nature Geoscience*, 9(12): 884–887. https://doi.org/10.1038/ngeo2837

**Reshmi, R.R., Deepa Nair, K., Zachariah, E.J. & Vincent, S.G.T.** 2015. Methanogenesis: Seasonal changes in human impacted regions of Ashtamudi estuary (Kerala, South India). *Estuarine, Coastal and Shelf Science*, 156: 144–154. https://doi.org/10.1016/j.ecss.2014.11.031

**Reuland, G., Sigurnjak, I., Dekker, H., Michels, E. & Meers, E.** 2021. The potential of digestate and the liquid fraction of digestate as chemical fertiliser substitutes under the RENURE criteria. *Agronomy*, 11: 1374. https://doi.org/10.3390/agronomy11071374

**Richa, K. & Ryen, E.G.** 2018. Policy landscape and recommendations to inform adoption of food waste-to-energy technologies. In: T.A. Trabold & C.W. Babbitt, eds. *Sustainable Food Waste-To-Energy Systems*. Elsevier Inc. https://doi.org/10.1016/C2016-0-00715-5

**Richardson, K., Steffen, W., Lucht, W., Bendtsen, J., Cornell, S.E., Donges, J.F., Drüke, M. *et al.*** 2023. Earth beyond six of nine planetary boundaries. *Science Advances*, 9(37): eadh2458. https://doi.org/10.1126/sciadv.adh2458

**Ricome, A., Barreiro-Hurle, J. & Sadibou Fall, C.** 2024. Government fertilizer subsidies, input use, and income: The case of Senegal. *Food Policy*, 124: 102623. https://doi.org/10.1016/j.foodpol.2024.102623

**Riley, W.J., Ortiz-Monasterio, I. & Matson, P.A.** 2001. Nitrogen leaching and soil nitrate, nitrite, and ammonium levels under irrigated wheat in Northern Mexico. *Nutrient Cycling in Agroecosystems*, 61(3): 223–236. https://doi.org/10.1023/A:1013758116346

**Rivera, J.E. & Chará, J.** 2021. $CH_4$ and $N_2O$ emissions from cattle excreta: A review of main drivers and mitigation strategies in grazing systems. *Frontiers in Sustainable Food Systems*, 5: 657936. https://doi.org/10.3389/fsufs.2021.657936

**RIVM (Rijksinstituut voor Volksgezondheid en Milieu).** 2022. *Toelichting bij richtinggevende emissiereductiedoelstellingen per gebied*. Memo, Rijksinstituut voor Volksgezondheid en Milieu. https://www.rivm.nl/sites/default/files/2022-06/RIVM-AERIUS_21-083_Toelichting%20bij%20richtinggevende%20emissiereductiedoelstellingen.pdf

**Rodríguez, A., Sanz-Cobeña, A., Ruiz-Ramos, M., Aguilera, E., Quemada, M., Billen, G., Garnier, J. & Lassaletta, L.** 2023. Nesting nitrogen budgets through spatial and system scales in the Spanish agro-food system over 26 years. *Science of The Total Environment*, 892: 164467. https://doi.org/10.1016/j.scitotenv.2023.164467

**Romer, P.S., Duffey, K.C., Wooldridge, P.J., Edgerton, E., Baumann, K., Feiner, P.A., Miller, D.O. *et al.*** 2018. Effects of temperature-dependent $NO_x$ emissions on continental ozone production. *Atmospheric Chemistry and Physics*, 18(4): 2601–2614. https://doi.org/10.5194/acp-18-2601-2018

**Rose, T.J., Wood, R.H., Rose, M.T. & van Zwieten, L.** 2018. A re-evaluation of the agronomic effectiveness of the nitrification inhibitors DCD and DMPP and the urease inhibitor NBPT. *Agriculture, Ecosystems & Environment*, 252: 69–73. https://doi.org/10.1016/j.agee.2017.10.008

**Roy, S.S., Rahman, A., Ahmed, S., Shahfahad & Ahmad, I.A.** 2020. Alarming groundwater depletion in the Delhi metropolitan region: A long-term assessment. *Environmental Monitoring and Assessment*, 192(10): 620. https://doi.org/10.1007/s10661-020-08585-8

**Rueda, B., McRoberts, K., Blake, R., Nicholson, C., Valentim, J. & Fernandes, E.** 2020. Nutrient status of cattle grazing systems in the western Brazilian Amazon. *Cogent Food & Agriculture*, 6(1): 1722350. https://doi.org/10.1080/23311932.2020.1722350

**Rufino, M.C., Tittonell, P., van Wijk, M., Castellanos-Navarrete, A., Delve, R., de Ridder, N. & Giller, K.** 2007. Manure as a key resource within smallholder farming systems: Analysing farm-scale nutrient cycling efficiencies with the NUANCES framework. *Livestock Science*, 112(3): 273–287. https://doi.org/10.1016/j.livsci.2007.09.011

**Russelle, M.P., Entz, M.H. & Franzluebbers, A.J.** 2007. Reconsidering integrated crop-livestock systems in North America. *Agronomy Journal*, 99(2): 325–334. https://doi.org/10.2134/agronj2006.0139

**Sajeev, E.P., Winiwarter, W. & Amon, B.** 2018. Greenhouse gas and ammonia emissions from different stages of liquid manure management chains: Abatement options and emission interactions. *Journal of Environmental Quality*, 47(1): 30–41. https://doi.org/10.2134/jeq2017.05.0199

**Sajeev, E.P.M., Amon, B., Ammon, C., Zollitsch, W. & Winiwarter, W.** 2018. Evaluating the potential of dietary crude protein manipulation in reducing ammonia emissions from cattle and pig manure: A meta-analysis. *Nutrient Cycling in Agroecosystems*, 110(1): 161–175. https://doi.org/10.1007/s10705-017-9893-3

**Saju, A., van de Sande, T., Ryan, D., Karpinska, A., Sigurnjak, I., Dowling, D.N., Germaine, K., Kakouli-Duarte, T. & Meers, E.** 2023. Exploring the short-term in-field performance of recovered nitrogen from manure (RENURE) materials to substitute synthetic nitrogen fertilisers. *Cleaner and Circular Bioeconomy*, 5: 100043. https://doi.org/10.1016/j.clcb.2023.100043

**Sakadevan, K. & Nguyen, M.-L.** 2017. Chapter Four: Livestock production and its impact on nutrient pollution and greenhouse gas emissions. In: D.L. Sparks, ed. *Advances in Agronomy*. pp. 147–184. Vol. 141. Academic Press. https://doi.org/10.1016/bs.agron.2016.10.002

**Sanchez, P.A.** 2002. Soil Fertility and Hunger in Africa. *Science*, 295(5562): 2019–2020. https://doi.org/10.1126/science.1065256

**Santagata, R., Ripa, M., Genovese, A. & Ulgiati, S.** 2021. Food waste recovery pathways: Challenges and opportunities for an emerging bio-based circular economy. A systematic review and an assessment. *Journal of Cleaner Production*, 286: 125490. https://doi.org/10.1016/j.jclepro.2020.125490

**Sanz-Cobena, A., Lassaletta, L., Aguilera, E., Del Prado, A., Garnier, J., Billen, G., Iglesias, A. *et al.*** 2017. Quantification and mitigation of greenhouse gas emissions in Mediterranean cropping systems, 238: 5–24. https://doi.org/10.1016/j.agee.2016.09.038

**Sapkota, T.B., Bijay-Singh & Takele, R.** 2023. Improving nitrogen use efficiency and reducing nitrogen surplus through best fertilizer nitrogen management in cereal production: The case of India and China. In: *Advances in Agronomy*. pp. 233–294. Vol. 178. Elsevier. https://doi.org/10.1016/bs.agron.2022.11.006

**Saud, S., Wang, D. & Fahad, S.** 2022. Improved nitrogen use efficiency and greenhouse gas emissions in agricultural soils as producers of biological nitrification inhibitors. *Frontiers in Plant Science*, 13. https://www.frontiersin.org/journals/plant-science/articles/10.3389/fpls.2022.854195

**Savin, R., Sadras, V.O. & Slafer, G.A.** 2019. Bench-marking nitrogen utilisation efficiency in wheat for Mediterranean and non-Mediterranean European regions. *Field Crops Research*, 241: 107573. https://doi.org/10.1016/j.fcr.2019.107573

**Schiere, J.B., Ibrahim, M. & van Keulen, H.** 2002. The role of livestock for sustainability in mixed farming: Criteria and scenario studies under varying resource allocation. *Agriculture, Ecosystems & Environment*, 90(2): 139–153. https://doi.org/10.1016/S0167-8809(01)00176-1

**Schirmer, J., Dovers, S. & Clayton, H.** 2012. Informing conservation policy design through an examination of landholder preferences: A case study of scattered tree conservation in Australia. *Biological Conservation*, 153: 51–63. https://doi.org/10.1016/j.biocon.2012.04.014

**Schlesinger, W.H.** 2009. On the fate of anthropogenic nitrogen. *Proceedings of the National Academy of Sciences*, 106(1): 203–208. https://doi.org/10.1073/pnas.0810193105

**Schrade, S., Zeyer, K., Mohn, J. & Zähner, M.** 2023. Effect of diets with different crude protein levels on ammonia and greenhouse gas emissions from a naturally ventilated dairy housing. *Science of the Total Environment*, 896: 165027. https://doi.org/10.1016/j.scitotenv.2023.165027

**Schulte-Uebbing, L. & de Vries, W.** 2021. Reconciling food production and environmental boundaries for nitrogen in the European Union. *Science of the Total Environment*, 786: 147427. https://doi.org/10.1016/j.scitotenv.2021.147427

**Schut, A.G.T., Cooledge, E.C., Moraine, M., van de Ven, G.W.J., Jones, D.L. & Chadwick, D.R.** 2021. Reintegration of crop-livestock systems in Europe: An overview. *Frontiers of Agricultural Science and Engineering*, 8(1): 111. https://doi.org/10.15302/J-FASE-2020373

**Schütz, L., Gattinger, A., Meier, M., Müller, A., Boller, T., Mäder, P. & Mathimaran, N.** 2018. Improving crop yield and nutrient use efficiency via biofertilization: A global meta-analysis. *Frontiers in Plant Science*, 8: 2204. https://doi.org/10.3389/fpls.2017.02204

**Sekaran, U., Lai, L., Ussiri, D.A.N., Kumar, S. & Clay, S.** 2021. Role of integrated crop-livestock systems in improving agriculture production and addressing food security: A review. *Journal of Agriculture and Food Research*, 5: 100190. https://doi.org/10.1016/j.jafr.2021.100190

**Serra, J., Medinets, S., Lassaletta, L., Zhang, X., Boincean, B. & Aguilera, E.** (forthcoming). Missing inputs and outputs. In: L. Lassaletta & A. Sanz-Cobeña, eds. *Guidance Document on nitrogen use efficiency indicators across multiple scales*. INMS *Guidance Document Series*. Edinburgh, UK Centre for Ecology & Hydrology.

**Sha, Z., Ma, X., Wang, J., Lv, T., Li, Q., Misselbrook, T. & Liu, X.** 2020. Effect of N stabilizers on fertilizer-N fate in the soil-crop system: A meta-analysis. *Agriculture, Ecosystems & Environment*, 290: 106763. https://doi.org/10.1016/j.agee.2019.106763

**Sha, Z., Liu, H., Wang, J., Ma, X., Liu, X. & Misselbrook, T.** 2021. Improved soil-crop system management aids in $NH_3$ emission mitigation in China. *Environmental Pollution*, 289: 117844. https://doi.org/10.1016/j.envpol.2021.117844

**Shao, S., Burns, D.A., Shen, H., Chen, Y., Russell, A.G. & Driscoll, C.T.** 2021. The response of streams in the Adirondack region of New York to projected changes in sulphur and nitrogen deposition under a changing climate. *Science of the Total Environment*, 800: 149626. https://doi.org/10.1016/j.scitotenv.2021.149626

**Shinde, R., Shahi, D.K., Mahapatra, P., Singh, C.S., Naik, S.K., Thombare, N. & Singh, A.K.** 2022. Management of crop residues with special reference to the on-farm utilization methods: A review. *Industrial Crops and Products*, 181: 114772. https://doi.org/10.1016/j.indcrop.2022.114772

**Shukla, S. & Saxena, A.** 2018. Global status of nitrate contamination in groundwater: Its occurrence, health impacts and mitigation measures. In: C.M. Hussain, ed. *Handbook of Environmental Materials Management*. pp. 1–21. Cham, Springer International Publishing. https://doi.org/10.1007/978-3-319-58538-3_20-1

**Shurson, G.C.** 2020. 'What a waste': Can we improve sustainability of food animal production systems by recycling food waste streams into animal feed in an era of health, climate, and economic crises? *Sustainability*, 12(17). https://doi.org/10.3390/su12177071

**SIAP (Servicio de Información agroalimentaria y Pesquera)**. 2023. *Panorama Agroalimentario 2023. Agricultura regenerativa, la vía para un futuro sustentable. Agricultura Secretaria de Agricultura y Desarrollo Rural, Servicio de Información Agroalimentaria y Pesquera de México*. México. https://drive.google.com/file/d/1FWHntHMgjw_uOse_MsOF9jZQDAm_FOD9/view

**Sigurdarson, J.J., Svane, S. & Karring, H.** 2018. The molecular processes of urea hydrolysis in relation to ammonia emissions from agriculture. *Reviews in Environmental Science and Bio/Technology*, 17(2): 241–258. https://doi.org/10.1007/s11157-018-9466-1

**Sileshi, G.W., Mafongoya, P., Akinnifesi, F., Phiri, E., Chirwa, P., Beedy, T., Makumba, W. *et al.*** 2014. Agroforestry: Fertilizer Trees. In: N.K. van Alfen, ed. *Encyclopedia of Agriculture and Food Systems.* Academic Press, Elsevier Inc.. http://dx.doi.org/10.1016/B978-0-444-52512-3.00022-X

**Sileshi, G.W., Kihara, J., Tamene, L., Vanlauwe, B., Phiri, E. & Jama, B.** 2022. Unravelling causes of poor crop response to applied N and P fertilizers on African soils. *Experimental Agriculture*, 58: e7. https://doi.org/10.1017/S0014479721000247

**Simon, W.J., Hijbeek, R., Frehner, A., Cardinaals, R., Talsma, E.F. & van Zanten, H.H.E.** 2024. Circular food system approaches can support current European protein intake levels while reducing land use and greenhouse gas emissions. *Nature Food*, 5(5): 402–412. https://doi.org/10.1038/s43016-024-00975-2

**Sinclair, T.R. & Rufty, T.W.** 2012. Nitrogen and water resources commonly limit crop yield increases, not necessarily plant genetics. *Global Food Security*, 1(2): 94–98. https://doi.org/10.1016/j.gfs.2012.07.001

**Singh, B.** 2016. *Agronomic benefits of neem coated urea – A review*. Review Papers. International Fertilizer Association. https://doi.org/10.13140/RG.2.2.10647.98722

**Sitko, N.J., Scognamillo, A. & Malevolti, G.** 2021. Does receiving food aid influence the adoption of climate-adaptive agricultural practices? Evidence from Ethiopia and Malawi. *Food Policy*, 102(1): 102041. https://doi.org/10.1016/j.foodpol.2021.102041

**Smith, V.H.** 2003. Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research*, 10(2): 126–139. https://doi.org/10.1065/espr2002.12.142

**Smith, J., Yeluripati, J., Smith, P. & Nayak, D.R.** 2020. Potential yield challenges to scale-up of zero budget natural farming. *Nature Sustainability*, 3(3): 247–252. https://doi.org/10.1038/s41893-019-0469-x

**Smits, M. & Montety, I.G.** 2009. Ammonia emission from camel dairy in the Netherlands. *Journal of Camel Practice and Research*, 16(2): 139–142. [Cited 15 January 2024]. https://www.camelsandcamelids.com/uploads/journal-manuscript/PG%20139-142%20Ammonia%20emission%20from.pdf

**Snapp, S., Sapkota, T.B., Chamberlin, J., Cox, C.M., Gameda, S., Jat, M.L., Marenya, P. *et al.*** 2023. Spatially differentiated nitrogen supply is key in a global food–fertilizer price crisis. *Nature Sustainability*, 6(10): 1268–1278. http://dx.doi.org/10.1038/s41893-023-01166-w

**Sobota, D.J., Compton, J.E., McCrackin, M.L. & Singh, S.** 2015. Cost of reactive nitrogen release from human activities to the environment in the United States. *Environmental Research Letters*, 10(2): 025006. https://doi.org/10.1088/1748-9326/10/2/025006

**Sommer, S. & Knudsen, L.** 2021. Impact of Danish livestock and manure management regulations on nitrogen pollution, crop production and economy. *Frontiers in Sustainability*, 2. https://doi.org/10.3389/frsus.2021.658231

**Sommer, S.G., Oenema, O., Matsunaka, T. & Jensen, L.S.** 2013. Regulations on animal manure management. In: *Animal Manure Recycling*. pp. 25–40. https://doi.org/10.1002/9781118676677.ch3

**Spiegal, S., Kleinman, P.J.A., Endale, D.M., Bryant, R.B., Dell, C., Goslee, S., Meinen, R.J. *et al.*** 2020. Manuresheds: Advancing nutrient recycling in US agriculture. *Agricultural Systems*, 182: 102813. https://doi.org/10.1016/j.agsy.2020.102813

**Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B.L., Lassaletta, L., de Vries, W. *et al.*** 2018. Options for keeping the food system within environmental limits. *Nature*, 562(7728): 519–525. https://doi.org/10.1038/s41586-018-0594-0

**Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R. *et al.*** 2015. Planetary boundaries: Guiding human development on a changing planet. *Science*, 347(6223): 1259855. https://doi.org/10.1126/science.1259855

**Steinfeld, H., Gerber, P., Wassenaar, T.D., Castel, V. & de Haan, C.** 2006. Livestock's long shadow: environmental issues and options. Food and Agriculture Organization.

**Stockmann, U., Adams, M.A., Crawford, J.W., Field, D.J., Henakaarchchi, N., Jenkins, M., Minasny, B. *et al.*** 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. *Agriculture, Ecosystems & Environment*, 164: 80–99. https://doi.org/10.1016/j.agee.2012.10.001

**Storkey, J., Macdonald, A.J., Poulton, P.R., Scott, T., Köhler, I.H., Schnyder, H., Goulding, K.W.T. & Crawley, M.J.** 2015. Grassland biodiversity bounces back from long-term nitrogen addition. *Nature*, 528(7582): 401–404. https://doi.org/10.1038/nature16444

**Strawn, D.G., Hinrich, L.B. & O´Connor, G.A.** 2020. *Soil Chemistry.* Wiley-Blackwell.

**Su, H., Wu, Y., Xia, W., Yang, L., Chen, J., Han, W., Fang, J. & Xie, P.** 2019. Stoichiometric mechanisms of regime shifts in freshwater ecosystem. *Water Research*, 149: 302–310. https://doi.org/10.1016/j.watres.2018.11.024

**Sugiura, K., Yamatani, S., Watahara, M. & Onodera, T.** 2009. Ecofeed, animal feed produced from recycled food waste. *Veterinaria Italiana*, 45: 397–404. https://pubmed.ncbi.nlm.nih.gov/20391403/

**Suh, S. & Yee, S.** 2011. Phosphorus use-efficiency of agriculture and food system in the US. *The Phosphorus Cycle*, 84(6): 806–813. https://doi.org/10.1016/j.chemosphere.2011.01.051

**Sumner, M.E. & Noble, A.D.** 2003. Soil acidification: The world story. In: Z. Rengel, ed. *Handbook of Soil Acidity*. New York, CRC Press. https://doi.org/10.1201/9780203912317

**Sun, B., Bai, Z., Li, Y., Li, R., Song, M., Xu, S., Zhang, H. & Zhuang, X.** 2022. Emission mitigation of $CH_4$ and $N_2O$ during semi-permeable membrane covered hyperthermophilic aerobic composting of livestock manure. *Journal of Cleaner Production*, 379: 134850. https://doi.org/10.1016/j.jclepro.2022.134850

**Sunday, J.M., Fabricius, K.E., Kroeker, K.J., Anderson, K.M., Brown, N.E., Barry, J.P., Connell, S.D. *et al.*** 2017. Ocean acidification can mediate biodiversity shifts by changing biogenic habitat. *Nature Climate Change*, 7(1): 81–85. https://doi.org/10.1038/nclimate3161

**Suri, T. & Udry, C.** 2022. Agricultural technology in Africa. *Journal of Economic Perspectives*, 36(1): 33–56. http://doi.org/10.1257/jep.36.1.33

**Sutton, M.A., Howard, C.M., Erisman, J.W., Billen, G., Bleeker, A., Grennfelt, P., van Grinsven, H. & Grizzetti, B.** 2011. *The European nitrogen assessment: Sources, Effects and Policy Perspectives*. Cambridge University Press. https://doi.org/10.1017/CBO9780511976988

**Sutton, M.A., Bleeker, A., Howard, C., Bekunda, M., Grizzetti, B., de Vries, W., van Grinsven, H. *et al.*** 2013. *Our Nutrient World: the challenge to produce more food and energy with less pollution.* Global Overview of Nutrient Management. Edinburgh, Centre for Ecology and Hydrology. https://nora.nerc.ac.uk/id/eprint/500700/1/N500700BK.pdf

**Sutton, M.A., Howard, C.M., Kanter, D.R., Lassaletta, L., Móring, A., Raghuram, N. & Read, N.** 2021. The nitrogen decade: mobilizing global action on nitrogen to 2030 and beyond. *One Earth*, 4(1): 10–14. http://dx.doi.org/10.1016/j.oneear.2020.12.016

**Sutton, M., Howard, C., Mason, K., Brownlie, W. & Cordovil, C.** 2022. *Nitrogen opportunities for agriculture, food & environment.* UNECE guidance document on integrated sustainable nitrogen management. UK Centre for Ecology & Hydrology. https://unece.org/sites/default/files/2022-11/UNECE_NitroOpps%20red.pdf

**Takata, M., Fukushima, K., Kino-Kimata, N., Nagao, N., Niwa, C. & Toda, T.** 2012. The effects of recycling loops in food waste management in Japan: based on the environmental and economic evaluation of food recycling. *Science of the Total Environment*, 432: 309–317. https://doi.org/10.1016/j.scitotenv.2012.05.049

**Tamagno, S., Maaz, T.M., van Kessel, C., Linquist, B.A., Ladha, J.K., Lundy, M.E., Maureira, F. & Pittelkow, C.M.** 2024. Critical assessment of nitrogen use efficiency indicators: Bridging new and old paradigms to improve sustainable nitrogen management. *European Journal of Agronomy*, 159: 127231. https://doi.org/10.1016/j.eja.2024.127231

**Tan, M., Hou, Y., Zhang, L., Shi, S., Long, W., Ma, Y., Zhang, T. & Oenema, O.** 2023. Decision-making environment of low-protein animal feeding in dairy and poultry farms in China. *Nutrient Cycling in Agroecosystems*, 127(1): 85–96. https://doi.org/10.1007/s10705-023-10295-9

**Tang, C. & Rengel, Z.** 2003. Role of Plant Cation/Anion Uptake Ratio in Soil Acidification. In: Z. Rengel, ed. *Handbook of Soil Acidity*. New York, CRC Press. https://doi.org/10.1201/9780203912317

**Teenstra, E., Vellinga, T., Aektasaeng, N., Amatayakul, W., Ndambi, A., Pelster, D., Germer, L., Jenet, A. & Andeweg, K.** 2014. *Global Assessment of Manure Management Policies and Practices*. Report 844, Wageningen, Wageningen Livestock Research. https://doi.org/10.6084/m9.figshare.8251232.v1

**Teutscherová, N., Vázquez, E., Sotelo, M., Villegas, D., Velásquez, N., Baquero, D., Pulleman, M. & Arango, J.** 2021. Intensive short-duration rotational grazing is associated with improved soil quality within one year after establishment in Colombia. *Applied Soil Ecology*, 159: 103835. https://doi.org/10.1016/j.apsoil.2020.103835

**Tewatia, R. & Chanda, T.** 2017. Trends in fertilizer nitrogen production and consumption in India. In: Y.P. Abrol, T.K. Adhya, V.P. Aneja, N. Raghuram, H. Pathak, U. Kulshrestha, C. Sharma & B. Singh, eds. *The Indian nitrogen assessment*. Elsevier Inc. https://doi.org/10.1016/B978-0-12-811836-8.00004-5

**Thakur, S., Chandel, R. & Narang, M.** 2018. Studies on Straw Management Techniques Using Paddy-Straw Chopper Cum Spreader Along with Various Tillage Practices and Subsequent Effect of Various Sowing Techniques on Wheat Yield and Economics. *Agricultural Mechanization In Asia, Africa And Latin America*, 49(2): 52–67.

**Thapa, R., Chatterjee, A., Awale, R., McGranahan, D.A. & Daigh, A.** 2016. Effect of Enhanced Efficiency Fertilizers on Nitrous Oxide Emissions and Crop Yields: A Meta-analysis. *Soil Science Society of America Journal*, 80(5): 1121–1134. https://doi.org/10.2136/sssaj2016.06.0179

**The Government of Brazil**. 2022. Decreto Nº 10.991, de 11 de Março 2022 – Institui o Plano Nacional de Fertilizantes 2022-2050 e o Conselho Nacional de Fertilizantes e Nutrição de Plantas. The Government of Brazil. https://www.fao.org/faolex/results/details/en/c/LEX-FAOC216335/

**The United Republic of Tanzania**. 2021. *National Sample Census of Agriculture 2019/20. National Report*. Dodoma, Tanzania National Bureau of Statistics. https://www.nbs.go.tz/nbs/takwimu/Agriculture/2019-20_Agri_Census_Main_Report.pdf

**Tian, D. & Niu, S.** 2015. A global analysis of soil acidification caused by nitrogen addition. *Environmental Research Letters*, 10(2): 024019. https://doi.org/10.1088/1748-9326/10/2/024019

**Tian, H., Xu, R., Canadell, J.G., Thompson, R.L., Winiwarter, W., Suntharalingam, P., Davidson, E.A. *et al.*** 2020. A comprehensive quantification of global nitrous oxide sources and sinks. *Nature*, 586(7828): 248–256. https://doi.org/10.1038/s41586-020-2780-0

**Tian, H., Bian, Z., Shi, H., Qin, X., Pan, N., Lu, C., Pan, S. *et al.*** 2022. History of anthropogenic Nitrogen inputs (HaNi) to the terrestrial biosphere: a 5 arcmin resolution annual dataset from 1860 to 2019. *Earth System Science Data*, 14(10): 4551-4568. https://doi.org/10.5194/essd-14-4551-2022

**Tittonell, P., Rufino, M.C., Janssen, B.H. & Giller, K.E.** 2010. Carbon and nutrient losses during manure storage under traditional and improved practices in smallholder crop-livestock systems – evidence from Kenya. *Plant and soil*, 328: 253–269. https://doi.org/10.1007/s11104-009-0107-x

**Tomich, T.P., Brodt, S.B., Dahlgren, R.A. & Scow, K.M. (eds.).** 2016. *The California nitrogen assessment: Challenges and solutions for people, agriculture, and the environment*. Oakland, University of California Press.

**Tosi, M., Mitter, E.K., Gaiero, J. & Dunfield, K.** 2020. It takes three to tango: the importance of microbes, host plant, and soil management to elucidate manipulation strategies for the plant microbiome. *Canadian Journal of Microbiology*, 66(7): 413–433. https://doi.org/10.1139/cjm-2020-0085

**Treseder, K.K.** 2008. Nitrogen additions and microbial biomass: a meta-analysis of ecosystem studies. *Ecology Letters*, 11(10): 1111–1120. https://doi.org/10.1111/j.1461-0248.2008.01230.x

**Tripathi, S., Srivastava, P., Devi, R.S. & Bhadouria, R.** 2020. Chapter 2 - Influence of synthetic fertilizers and pesticides on soil health and soil microbiology. In: M.N.V. Prasad, ed. *Agrochemicals Detection, Treatment and Remediation*. pp. 25–54. Butterworth-Heinemann. https://doi.org/10.1016/B978-0-08-103017-2.00002-7

**Tullo, E., Finzi, A. & Guarino, M.** 2019. Review: Environmental impact of livestock farming and Precision Livestock Farming as a mitigation strategy. *Science of The Total Environment*, 650: 2751–2760. https://doi.org/10.1016/j.scitotenv.2018.10.018

**Ulloa-Murillo, L.M., Villegas, L.M., Rodríguez-Ortiz, A.R., Duque-Acevedo, M. & Cortés-García, F.J.** 2022. Management of the organic fraction of municipal solid waste in the context of a sustainable and circular model: Analysis of trends in Latin America and the Caribbean. *International Journal of Environmental Research and Public Health*, 19(10): 6041. https://doi.org/10.3390/ijerph19106041

**UN (United Nations)**. 2024. SDG 2: End hunger, achieve food security and improved nutrition and promote sustainable agriculture. In: *Sustainable Development Goals*. [Cited 4 August 2024]. https://sdgs.un.org/goals/goal2

**UN**. 2023. *The Sustainable Development Goals Report 2023: Special Edition. Towards a Rescue Plan for People and Planet*. New York, United Nations Department of Economic and Social Affairs. [cited 20 January 2024]. https://unstats.un.org/sdgs/report/2023/The-Sustainable-Development-Goals-Report-2023.pdf

**UNCCD (United Nations Convention to Combat Desertification)**. 2024. Global Land Outlook Thematic Report on Rangelands and Pastoralism. United Nations Convention to Combat Desertification. https://www.unccd.int/sites/default/files/2024-05/GLO%20rangelands%20summary.pdf

**UNECE**. 1999. *The 1999 Gothenburg Protocol to abate acidification, eutrophication and ground-level ozone*. Convention on Long-range Transboundary Air Pollution, The United Nations Economic Commission for Europe, 1999. https://unece.org/environment-policy/air/protocol-abate-acidification-eutrophication-and-ground-level-ozone

**UNECE**. 2010. *ECE/EB.AIR/WG.5/2010/13. Options for revising the 1999 Gothenburg Protocol to abate acidification, eutrophication and ground-level ozone*. executive body for the convention on long-range transboundary air pollution, The United Nations Economic Commission for Europe, 2010. https://unece.org/fileadmin/DAM/env/documents/2010/eb/wg5/wg47/ECE.EB.AIR.WG.5.2010.13_e.pdf

**UNEP (United Nations Environment Programme)-WCMC (World Conservation Monitoring Centre)**. 2010. *Ecosystems and Human Well-Being: A Manual for Assessment Practitioners*. Washington, DC, Island Press. https://wedocs.unep.org/20.500.11822/8949

**UNEP**. 2019a. Resolution on Sustainable Nitrogen Management UNEP/EA.4/L.16. https://wedocs.unep.org/bitstream/handle/20.500.11822/28478/English.pdf

**UNEP**. 2019b. Colombo Declaration calls for tackling global nitrogen challenge. [Cited 5 May 2024]. https://www.unep.org/news-and-stories/press-release/colombo-declaration-calls-tackling-global-nitrogen-challenge

**UNEP**. 2022. UNEP/EA.5/Res.2. Resolution adopted by the United Nations Environment Assembly on 2 March 2022. 5/2 Sustainable nitrogen management. [Cited 5 May 2024]. https://wedocs.unep.org/bitstream/handle/20.500.11822/39816/SUSTAINABLE%20NITROGEN%20MANAGEMENT.%20English.pdf?sequence=1&isAllowed=y

**UNEP**. 2024. Food Waste Index Report 2024. United Nations Environment Programme.

**UNFCCC (United Nations Framework Convention on Climate Change)**. 2022. *What is the Triple Planetary Crisis?* [Cited 15 May 2024]. https://unfccc.int/news/what-is-the-triple-planetary-crisis

**Upadhaya, S., Arbuckle, J.G. & Schulte, L.A.** 2023. Individual- and county-level factors associated with farmers' use of 4R Plus nutrient management practices. *Journal of Soil and Water Conservation*, 78(5): 412. https://doi.org/10.2489/jswc.2023.00002

**US EPA (United States Environmental Protection Agency).** 2015. Preamble to the Integrated Science Assessments (ISA). [Cited 7 May 2024]. https://cfpub.epa.gov/ncea/isa/recordisplay.cfm?deid=310244

**US EPA.** 2016. Climate Change Indicators: Climate Forcing. [Cited 24 April 2024]. https://www.epa.gov/climate-indicators/climate-change-indicators-climate-forcing

**US EPA**. 2024. United States 2030 food loss and waste reduction goal. In: *Sustainable Management of Food*. [Cited 22 July 2024]. https://www.epa.gov/sustainable-management-food/united-states-2030-food-loss-and-waste-reduction-goal

**Uwizeye, A.** 2019. *Nutrient challenges in global livestock supply chains : an assessment of nitrogen use and flows*. Wageningen University. https://doi.org/10.18174/469578

**Uwizeye, A., Gerber, P.J., Schulte, R.P.O. & de Boer, I.J.M.** 2016. A comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains. *Journal of Cleaner Production*, 129: 647–658. https://doi.org/10.1016/j.jclepro.2016.03.108

**Uwizeye, A., Gerber, P.J., Opio, C.I., Tempio, G., Mottet, A., Makkar, H.P.S., Falcucci, A., Steinfeld, H. & de Boer, I.J.M.** 2019. Nitrogen flows in global pork supply chains and potential improvement from feeding swill to pigs. *Resources, Conservation and Recycling*, 146: 168–179. https://doi.org/10.1016/j.resconrec.2019.03.032

**Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

**Vachon, D., Sadro, S., Bogard, M.J., Lapierre, J.-F., Baulch, H.M., Rusak, J.A., Denfeld, B.A. *et al.*** 2020. Paired $O_2$–$CO_2$ measurements provide emergent insights into aquatic ecosystem function. *Limnology and Oceanography Letters*, 5(4): 287–294. https://doi.org/10.1002/lol2.10135

**Valenzuela, H.** 2023. Ecological Management of the Nitrogen Cycle in Organic Farms. *Nitrogen*, 4(1): 58–84. https://doi.org/10.3390/nitrogen4010006

**Valve, H., Ekholm, P. & Luostarinen, S.** 2020. The circular nutrient economy: needs and potentials of nutrient recycling. In: M. Brandão, D. Lazarevic & G. Finnveden, eds. *Handbook of the Circular Economy.* Edward Elgar Publishing. https://doi.org/10.4337/9781788972727.00037

**van Egmond, K., Bresser, T. & Bouwman, L.** 2002. The European Nitrogen Case. *AMBIO: A Journal of the Human Environment*, 31(2): 72–78. https://doi.org/10.1579/0044-7447-31.2.72

**van Grinsven, H.J., Bouwman, L., Cassman, K.G., van Es, H.M., McCrackin, M.L. & Beusen, A.H.** 2015. Losses of ammonia and nitrate from agriculture and their effect on nitrogen recovery in the European Union and the United States between 1900 and 2050. *Journal of Environmental Quality*, 44(2): 356–367. https://doi.org/10.2134/jeq2014.03.0102

**van Grinsven, H., Ebanyat, P., Glendining, M., Gu, B., Hijbeek, R., Lam, S.K., Lassaletta, L. *et al.*** 2022. Establishing long-term nitrogen response of global cereals to assess sustainable fertilizer rates. *Nature Food*, 3(2): 122–132. https://doi.org/10.1038/s43016-021-00447-x

**van Groenigen, J.W., van Kessel, C., Hungate, B.A., Oenema, O., Powlson, D.S. & van Groenigen, K.J.** 2017. Sequestering Soil Organic Carbon: A Nitrogen Dilemma. *Environmental Science & Technology*, 51(9): 4738–4739. https://doi.org/10.1021/acs.est.7b01427

**van Hung, N., Detras, M.C., Migo, M., Quilloy, R., Balingbing, C., Chivenge, P. & Gummert, M.** 2020. Rice Straw Overview: Availability, Properties, and Management Practices. pp. 1–13. https://doi.org/10.1007/978-3-030-32373-8_1

**van Ittersum, M.K., van Bussel, L.G., Wolf, J., Grassini, P., van Wart, J., Guilpart, N., Claessens, L. *et al.*** 2016. Can sub-Saharan Africa feed itself? *Proceedings of the National Academy of Sciences*, 113(52): 14964–14969. https://doi.org/10.1073/pnas.1610359113

**van Kernebeek, H.R.J., Oosting, S.J., van Ittersum, M.K., Bikker, P. & de Boer, I.J.M.** 2016. Saving land to feed a growing population: consequences for consumption of crop and livestock products. *The International Journal of Life Cycle Assessment*, 21(5): 677–687. https://doi.org/10.1007/s11367-015-0923-6

**van Loon, M.P., Vonk, W.J., Hijbeek, R., van Ittersum, M.K. & ten Berge, H.F.** 2023. Circularity indicators and their relation with nutrient use efficiency in agriculture and food systems. *Agricultural Systems*, 207: 103610. https://doi.org/10.1016/j.agsy.2023.103610

**van Selm, B., Hijbeek, R., van Ittersum, M.K., van Hal, O., van Middelaar, C.E. & de Boer, I.J.M.** 2023. Recoupling livestock and feed production in the Netherlands to reduce environmental impacts. *Science of The Total Environment*, 899: 165540. https://doi.org/10.1016/j.scitotenv.2023.165540

**van Wesenbeeck, C., Keyzer, M., van Veen, W. & Qiu, H.** 2021. Can China's overuse of fertilizer be reduced without threatening food security and farm incomes? *Agricultural Systems*, 190: 103093. https://doi.org/10.1016/j.agsy.2021.103093

**van Zanten, H.H.E., van Ittersum, M.K. & de Boer, I.J.M.** 2019. The role of farm animals in a circular food system. *Global Food Security*, 21: 18–22. https://doi.org/10.1016/j.gfs.2019.06.003

**VanderZaag, A., Amon, B., Bittman, S. & Kuczyński, T.** 2015. Ammonia abatement with manure storage and processing techniques. In: S. Reis, C. Howard & M. Sutton, eds. *Costs of Ammonia Abatement and the Climate Co-Benefits*. Springer. http://dx.doi.org/10.1007/978-94-017-9722-1_5

**Vanham, D., Leip, A., Galli, A., Kastner, T., Bruckner, M., Uwizeye, A., van Dijk, K. *et al.*** 2019. Environmental footprint family to address local to planetary sustainability and deliver on the SDGs. *Science of The Total Environment*, 693: 133642. https://doi.org/10.1016/j.scitotenv.2019.133642

**Varol, S. & Şekerci, M.** 2018. Hydrogeochemistry, water quality and health risk assessment of water resources contaminated by agricultural activities in Korkuteli (Antalya, Turkey) district center. *Journal of Water and Health*, 16(4): 574–599. https://doi.org/10.2166/wh.2018.003

**Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H. & Tilman, D.G.** 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications*, 7(3): 737–750. https://doi.org/10.1890/1051-0761(1997)007[0737:HAOTGN]2.0.CO;2

**Vitousek, P.M., Porder, S., Houlton, B.Z. & Chadwick, O.A.** 2010. Terrestrial phosphorus limitation: mechanisms, implications, and nitrogen–phosphorus interactions. *Ecological Applications*, 20(1): 5–15. http://dx.doi.org/10.1890/08-0127.1

**Vonk, W.J., Hijbeek, R., Glendining, M.J., Powlson, D.S., Bhogal, A., Merbach, I., Silva, J.V. *et al.*** 2022. The legacy effect of synthetic N fertiliser. *European Journal of Soil Science*, 73(3): e13238. https://doi.org/10.1111/ejss.13238

**Waddington, S.R., Li, X., Dixon, J., Hyman, G. & De Vicente, M.C.** 2010. Getting the focus right: production constraints for six major food crops in Asian and African farming systems. *Food Security*, 2(1): 27–48. https://doi.org/10.1007/s12571-010-0053-8

**Wako, G., Tadesse, M. & Angassa, A.** 2017. Camel management as an adaptive strategy to climate change by pastoralists in southern Ethiopia. *Ecological Processes*, 6(1): 26. https://doi.org/10.1186/s13717-017-0093-5

**Wang, Q., Zhou, F., Shang, Z., Ciais, P., Winiwarter, W., Jackson, R.B., Tubiello, F.N. *et al.*** 2020. Data-driven estimates of global nitrous oxide emissions from croplands. *National Science Review*, 7(2): 441–452. https://doi.org/10.1093/nsr/nwz087

**Wang, J.M., Liu, Q., Hou, Y., Qin, W., Bai, Z.H., Zhang, F.S. & Oenema, O.** 2022a. Impacts of international food and feed trade on nitrogen balances and nitrogen use efficiencies of food systems. *Science of The Total Environment*, 838: 156151. https://doi.org/10.1016/j.scitotenv.2022.156151

**Wang, Y., de Boer, I.J.M., Hou, Y. & van Middelaar, C.E.** 2022b. Manure as waste and food as feed: Environmental challenges on Chinese dairy farms. *Resources, Conservation and Recycling*, 181: 106233. https://doi.org/10.1016/j.resconrec.2022.106233

**Wang, X., Xu, M., Lin, B., Bodirsky, B.L., Xuan, J., Dietrich, J.P., Stevanović, M. *et al.*** 2023. Reforming China's fertilizer policies: implications for nitrogen pollution reduction and food security. *Sustainability Science*, 18(1): 407–420. https://doi.org/10.1007/s11625-022-01189-w

**Wassen, M.J., Schrader, J., Eppinga, M.B., Sardans, J., Berendse, F., Beunen, R., Peñuelas, J. & van Dijk, J.** 2022. The EU needs a nutrient directive. *Nature Reviews Earth & Environment*, 3(5): 287–288. https://doi.org/10.1038/s43017-022-00295-8

**Watson, C.A. & Atkinson, D.** 1999. Using nitrogen budgets to indicate nitrogen use efficiency and losses from whole farm systems: a comparison of three methodological approaches. *Nutrient Cycling in Agroecosystems*, 53(3): 259–267. https://doi.org/10.1023/A:1009793120577

**Watson, C.A., Topp, C.F. & Ryschawy, J.** 2019. Linking arable cropping and livestock production for efficient recycling of N and P. In: G. Lemaire, P. Carvalho, S. Kronberg & S. Recous, eds. *Agroecosystem Diversity*. Academic Press, Elsevier. https://doi.org/10.1016/B978-0-12-811050-8.00010-8

**Wattiaux, M.A., Uddin, M.E., Letelier, P., Jackson, R.D. & Larson, R.A.** 2019. Invited Review: Emission and mitigation of greenhouse gases from dairy farms: The cow, the manure, and the field. *Applied Animal Science*, 35(2): 238–254. https://doi.org/10.15232/aas.2018-01803

**Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A. *et al.*** 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences*, 106(30): 12377–12381. https://doi.org/10.1073/pnas.0905620106

**WEF (World Economic Forum)**. 2020. *The Future Of Nature And Business*. New Nature Economy Report 2. Geneva, Switzerland, World Economic Forum. https://www3.weforum.org/docs/WEF_The_Future_Of_Nature_And_Business_2020.pdf

**Wei, S., Zhu, Z., Zhao, J., Chadwick, D.R. & Dong, H.** 2021. Policies and regulations for promoting manure management for sustainable livestock production in China: A review. *Frontiers of Agricultural Science and Engineering*, 8(1): 45–57. https://doi.org/10.15302/J-FASE-2020369

**Wei, Z., Zhuang, M., Hellegers, P., Cui, Z. & Hoffland, E.** 2023. Towards circular nitrogen use in the agri-food system at village and county level in China. *Agricultural Systems*, 209: 103683. https://doi.org/10.1016/j.agsy.2023.103683

**Weil, R. & Brady, N.** 2017. *The Nature and Properties of Soils. 15th edition.* Pearson Education.

**West, W.E., Creamer, K.P. & Jones, S.E.** 2016. Productivity and depth regulate lake contributions to atmospheric methane. *Limnology and Oceanography*, 61(S1). https://doi.org/10.1002/lno.10247

**Wick, K., Heumesser, C. & Schmid, E.** 2012. Groundwater nitrate contamination: Factors and indicators. *Journal of Environmental Management*, 111: 178–186. https://doi.org/10.1016/j.jenvman.2012.06.030

**Wiedemann, S., Phillips, F.A., Naylor, T.A., McGahan, E., Keane, O., Warren, B. & Murphy, C.** 2016. Nitrous oxide, ammonia and methane from Australian meat chicken houses measured under commercial operating conditions and with mitigation strategies applied. *Animal Production Science*, 56(9): 1404–1417. https://doi.org/10.1071/AN15561

**Wiesmeier, M., Urbanski, L., Hobley, E., Lang, B., von Lützow, M., Marin-Spiotta, E., van Wesemael, B. *et al.*** 2019. Soil organic carbon storage as a key function of soils – A review of drivers and indicators at various scales. *Geoderma*, 333: 149–162. https://doi.org/10.1016/j.geoderma.2018.07.026

**Wilkinson, J.** 2011. Re-defining efficiency of feed use by livestock. *Animal*, 5(7): 1014–1022. https://doi.org/10.1017/s175173111100005x

**Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T. *et al.*** 2019. Food in the Anthropocene: the EAT–Lancet Commission on healthy diets from sustainable food systems. *The Lancet*, 393(10170): 447–492. https://doi.org/10.1016/S0140-6736(18)31788-4

**Wise, T.A.** 2021. *Old fertilizer in new bottles: Selling the past as innovation in Africa's Green Revolution*. Working Paper No. 21-01. Medford, Global Development and Environment Institute. [Cited 19 April 2024] https://mronline.org/wp-content/uploads/2021/04/21-01Wise_OldFertilizer.pdf

**Wood, L., Lubell, M., Rudnick, J., Khalsa, S.D.S., Sears, M. & Brown, P.H.** 2022. Mandatory information-based policy tools facilitate California farmers' learning about nitrogen management. *Land Use Policy*, 114: 105923. https://doi.org/10.1016/j.landusepol.2021.105923

**Wu, L., Chen, X., Wei, W., Liu, Y., Wang, D. & Ni, B.-J.** 2020. A Critical Review on Nitrous Oxide Production by Ammonia-Oxidizing Archaea. *Environmental Science & Technology*, 54(15): 9175–9190. https://doi.org/10.1021/acs.est.0c03948

**Wu, Z., Feng, X., Zhang, Y. & Fan, S.** 2024. Repositioning fertilizer manufacturing subsidies for improving food security and reducing greenhouse gas emissions in China. *Journal of Integrative Agriculture*, 23(2): 430–443. https://doi.org/10.1016/j.jia.2023.12.007

**Wuebbles, D.** 2002. Atmospheric methane and global change. *Earth-Science Reviews*, 57(3–4): 177–210. https://doi.org/10.1016/S0012-8252(01)00062-9

**Xia, J. & Wan, S.** 2008. Global response patterns of terrestrial plant species to nitrogen addition. *New Phytologist*, 179(2): 428–439. https://doi.org/10.1111/j.1469-8137.2008.02488.x

**Xia, L., Lam, S.K., Chen, D., Wang, J., Tang, Q. & Yan, X.** 2017. Can knowledge-based N management produce more staple grain with lower greenhouse gas emission and reactive nitrogen pollution? A meta-analysis. *Global Change Biology*, 23(5): 1917–1925. https://doi.org/10.1111/gcb.13455

**Xu, S., Jagadamma, S. & Rowntree, J.** 2018. Response of Grazing Land Soil Health to Management Strategies: A Summary Review. *Sustainability*, 10(12): 4769. https://doi.org/10.3390/su10124769

**Xu, R., Tian, H., Pan, S., Dangal, S.R.S., Chen, J., Chang, J., Lu, Y. *et al.*** 2019. Increased nitrogen enrichment and shifted patterns in the world's grassland: 1860-2016. *Earth System Science Data*, 11(1): 175–187. https://doi.org/10.5194/essd-11-175-2019

**Yan, X., Xu, X., Wang, M., Wang, G., Wu, S., Li, Z., Sun, H., Shi, A. & Yang, Y.** 2017. Climate warming and cyanobacteria blooms: Looks at their relationships from a new perspective. *Water Research*, 125: 449–457. https://doi.org/10.1016/j.watres.2017.09.008

**Yan, M., Pan, G., Lavallee, J.M. & Conant, R.T.** 2020. Rethinking sources of nitrogen to cereal crops. *Global Change Biology*, 26(1): 191–199. https://doi.org/10.1111/gcb.14908

**Yang, Y., Li, Z. & Zhang, Y.** 2021. Incentives or restrictions: policy choices in farmers' chemical fertilizer reduction and substitution behaviors. *International Journal of Low-Carbon Technologies*, 16(2): 351–360. https://doi.org/10.1093/ijlct/ctaa068

**Yao, G., Zhang, X., Davidson, E.A. & Taheripour, F.** 2021. The increasing global environmental consequences of a weakening US–China crop trade relationship. *Nature Food*, 2(8): 578–586. https://doi.org/10.1038/s43016-021-00338-1

**You, L., Ros, G.H., Chen, Y., Shao, Q., Young, M.D., Zhang, F. & de Vries, W.** 2023. Global mean nitrogen recovery efficiency in croplands can be enhanced by optimal nutrient, crop and soil management practices. *Nature Communications*, 14(1): 5747. https://doi.org/10.1038/s41467-023-41504-2

**Yu, Q., Duan, L., Yu, L., Chen, X., Si, G., Ke, P., Ye, Z. & Mulder, J.** 2018. Threshold and multiple indicators for nitrogen saturation in subtropical forests. *Environmental Pollution*, 241: 664–673. https://doi.org/10.1016/j.envpol.2018.06.001

**Yu, Z., Jin, X., Miao, L. & Yang, X.** 2021. A historical reconstruction of cropland in China from 1900 to 2016. *Earth System Science Data*, 13(7): 3203–3218. https://doi.org/10.5194/essd-13-3203-2021

**Zarrin, M., Riveros, J.L., Ahmadpour, A., De Almeida, A.M., Konuspayeva, G., Vargas-Bello-Pérez, E., Faye, B. & Hernández-Castellano, L.E.** 2020. Camelids: new players in the international animal production context. *Tropical Animal Health and Production*, 52(3): 903–913. https://doi.org/10.1007/s11250-019-02197-2

**Zeri, G.C. & Ometto, J.P.** 2018. *Nitrogen emissions in Latin America: impacts, drivers, and policy responses*. Washington, AGU Fall Meeting 2018. http://mtc-m21c.sid.inpe.br/col/sid.inpe.br/mtc-m21c/2018/12.03.12.32/doc/AGU%202018_abstract%20455157_B13A-06_Gisleine%20CUNHA%20ZERI.pdf

**Zhang, J.** 2011. China's success in increasing per capita food production. *Journal of Experimental Botany*, 62(11): 3707–3711. https://doi.org/10.1093/jxb/err132

**Zhang, X.** 2017. Biogeochemistry: A plan for efficient use of nitrogen fertilizers. *Nature*, 543(7645): 322–323. https://doi.org/10.1038/543322a

**Zhang, Z. & Furman, A.** 2023. Statistical analysis for biogeochemical processes in a sandy column with dynamic hydrologic regimes using spectral induced polarization (SIP) and self-potential (SP). *Geophysical Journal International*, 233(1): 564–585. https://doi.org/10.1093/gji/ggac452

**Zhang, X., Davidson, E.A., Mauzerall, D.L., Searchinger, T.D., Dumas, P. & Shen, Y.** 2015a. Managing nitrogen for sustainable development. *Nature*, 528(7580): 51–59. https://doi.org/10.1038/nature15743

**Zhang, X., Mauzerall, D.L., Davidson, E.A., Kanter, D.R. & Cai, R.** 2015b. The Economic and Environmental Consequences of Implementing Nitrogen-Efficient Technologies and Management Practices in Agriculture. *Journal of Environmental Quality*, 44(2): 312–324. https://doi.org/10.2134/jeq2014.03.0129

**Zhang, Y., Li, C., Wang, Y., Hu, Y., Christie, P., Zhang, J. & Li, X.** 2016. Maize yield and soil fertility with combined use of compost and inorganic fertilizers on a calcareous soil on the North China Plain. *Soil and Tillage Research*, 155: 85–94. https://doi.org/10.1016/j.still.2015.08.006

**Zhang, C., Liu, S., Wu, S., Jin, S., Reis, S., Liu, H. & Gu, B.** 2019a. Rebuilding the linkage between livestock and cropland to mitigate agricultural pollution in China. *Resources, Conservation and Recycling*, 144: 65–73. https://doi.org/10.1016/j.resconrec.2019.01.011

**Zhang, J., Gao, Y., Leung, L.R., Luo, K., Liu, H., Lamarque, J.-F., Fan, J. *et al.*** 2019b. Impacts of climate change and emissions on atmospheric oxidized nitrogen deposition over East Asia. *Atmospheric Chemistry and Physics*, 19(2): 887–900. https://doi.org/10.5194/acp-19-887-2019

**Zhang, X., Davidson, E.A., Zou, T., Lassaletta, L., Quan, Z., Li, T. & Zhang, W.** 2020. Quantifying Nutrient Budgets for Sustainable Nutrient Management. *Global Biogeochemical Cycles*, 34(3): e2018GB006060. https://doi.org/10.1029/2018GB006060

**Zhang, X., Zou, T., Lassaletta, L., Mueller, N.D., Tubiello, F.N., Lisk, M.D., Lu, C. *et al.*** 2021. Quantification of global and national nitrogen budgets for crop production. *Nature Food*, 2(7): 529–540. https://doi.org/10.1038/s43016-021-00318-5

**Zhang, Y., Ye, C., Su, Y., Peng, W., Lu, R., Liu, Y., Huang, H. *et al.*** 2022. Soil Acidification caused by excessive application of nitrogen fertilizer aggravates soil-borne diseases: Evidence from literature review and field trials. *Agriculture, Ecosystems & Environment*, 340: 108176. https://doi.org/10.1016/j.agee.2022.108176

**Zheng, S.J.** 2010. Crop production on acidic soils: overcoming aluminium toxicity and phosphorus deficiency. *Annals of Botany*, 106(1): 183–184. https://doi.org/10.1093/aob/mcq134

**Zhou, L., Zhao, P., Chi, Y., Wang, D., Wang, P., Liu, N., Cai, D., Wu, Z. & Zhong, N.** 2017. Controlling the Hydrolysis and Loss of Nitrogen Fertilizer (Urea) by using a Nanocomposite Favors Plant Growth. *ChemSusChem*, 10(9): 2068–2079. https://doi.org/10.1002/cssc.201700032

**Zhou, Z., Zhang, S., Jiang, N., Xiu, W., Zhao, J. & Yang, D.** 2022. Effects of organic fertilizer incorporation practices on crops yield, soil quality, and soil fauna feeding activity in the wheat-maize rotation system. *Frontiers in Environmental Science*, 10. https://www.frontiersin.org/articles/10.3389/fenvs.2022.1058071

**Zhou, J., Zheng, Y., Hou, L., An, Z., Chen, F., Liu, B., Wu, L. *et al.*** 2023. Effects of acidification on nitrification and associated nitrous oxide emission in estuarine and coastal waters. *Nature Communications*, 14(1): 1380. https://doi.org/10.1038/s41467-023-37104-9

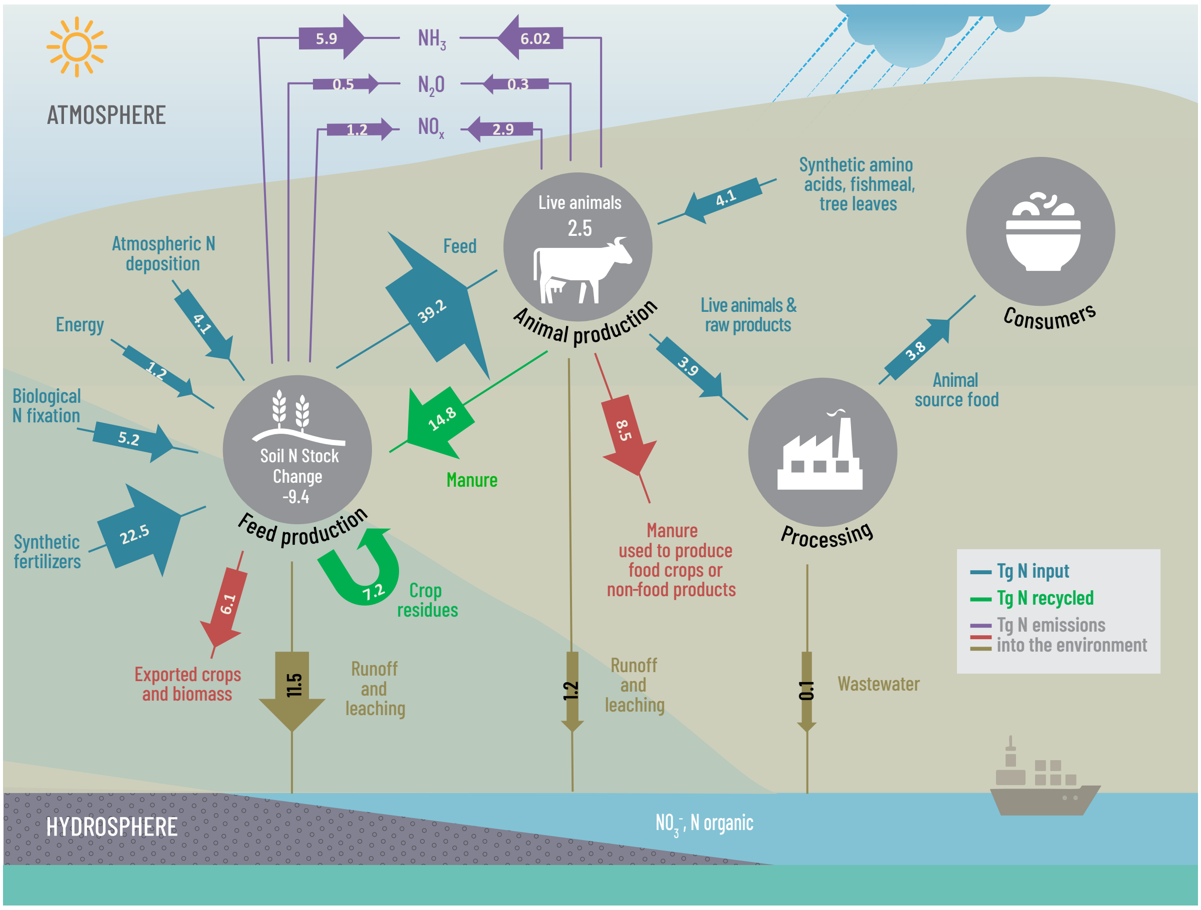
**Zhu, X., Ros, G.H., Xu, M., Cai, Z., Sun, N., Duan, Y. & de Vries, W.** 2023. Long-term impacts of mineral and organic fertilizer inputs on nitrogen use efficiency for different cropping systems and site conditions in Southern China. *European Journal of Agronomy*, 146: 126797. https://doi.org/10.1016/j.eja.2023.126797

**Zu Ermgassen, E., Phalan, B., Green, R. & Balmford, A.** 2016. Reducing the land use of EU pork production: Where there's swill, there's a way. *Food Policy*, 58: 35–48. https://doi.org/10.1016/j.foodpol.2015.11.001

## Annex

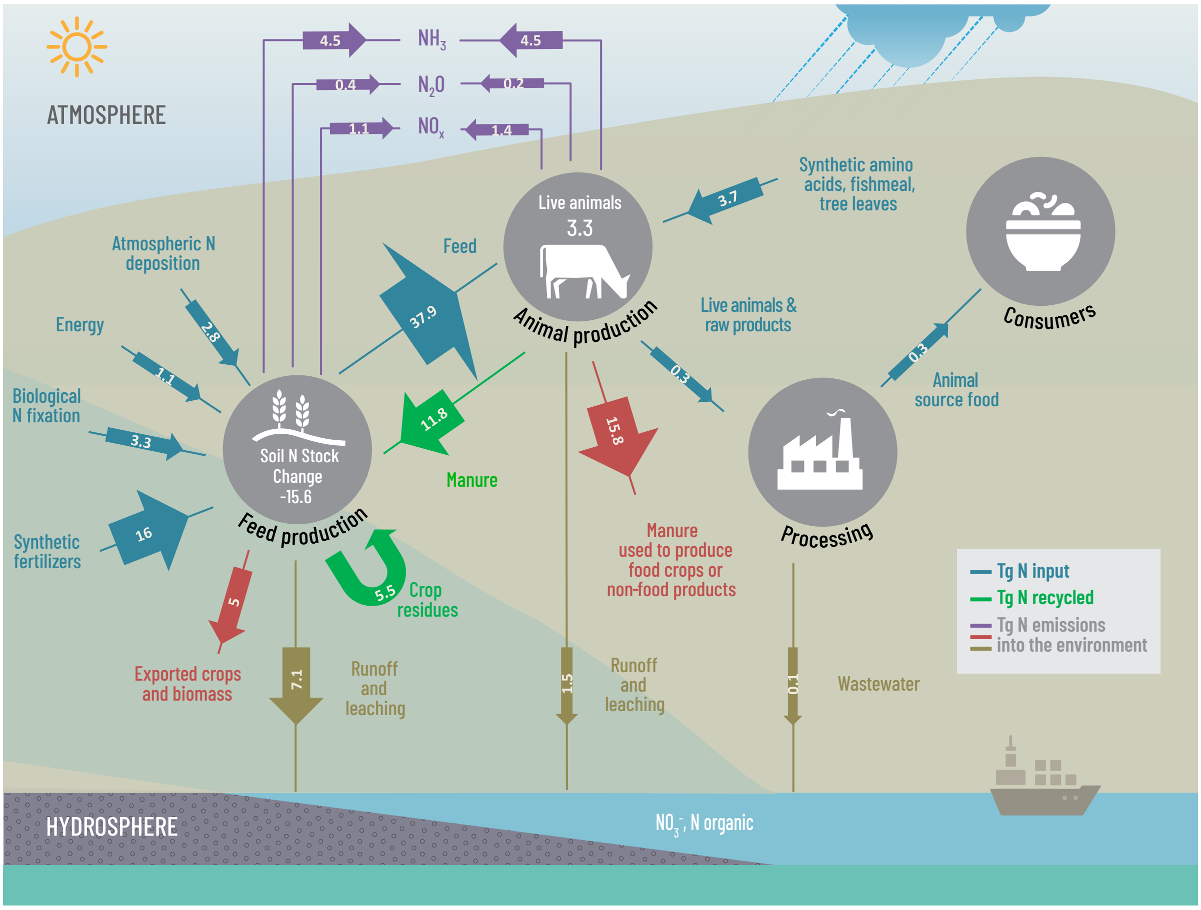
# Global nitrogen flows for main livestock species

FIGURE A.1
**Global N flows and sources of N compounds emissions allocated to dairy cattle for 2010 (Tg N/yr)**




*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

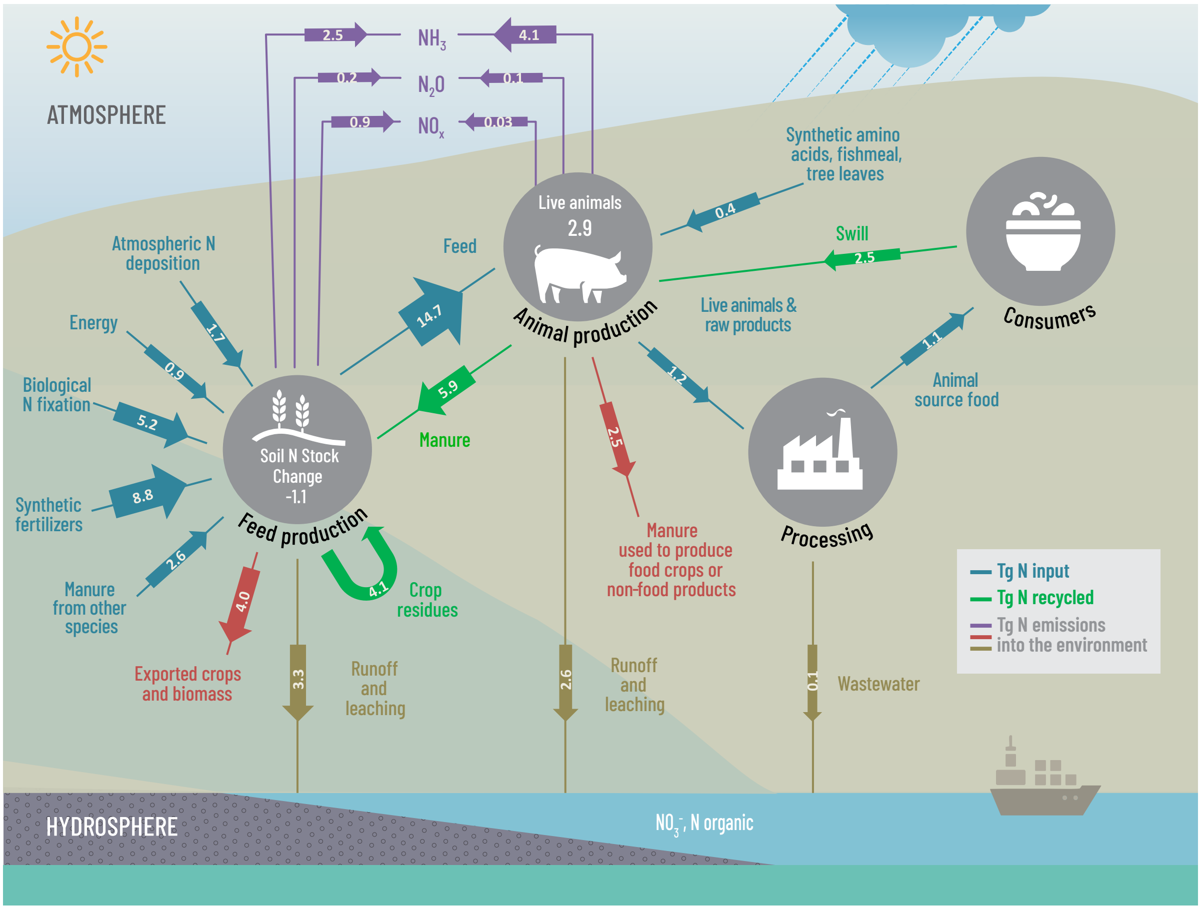
FIGURE A.2
**Global N flows and sources of N compounds emissions allocated to beef cattle for 2010 (Tg N/yr)**




*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F.** ***et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

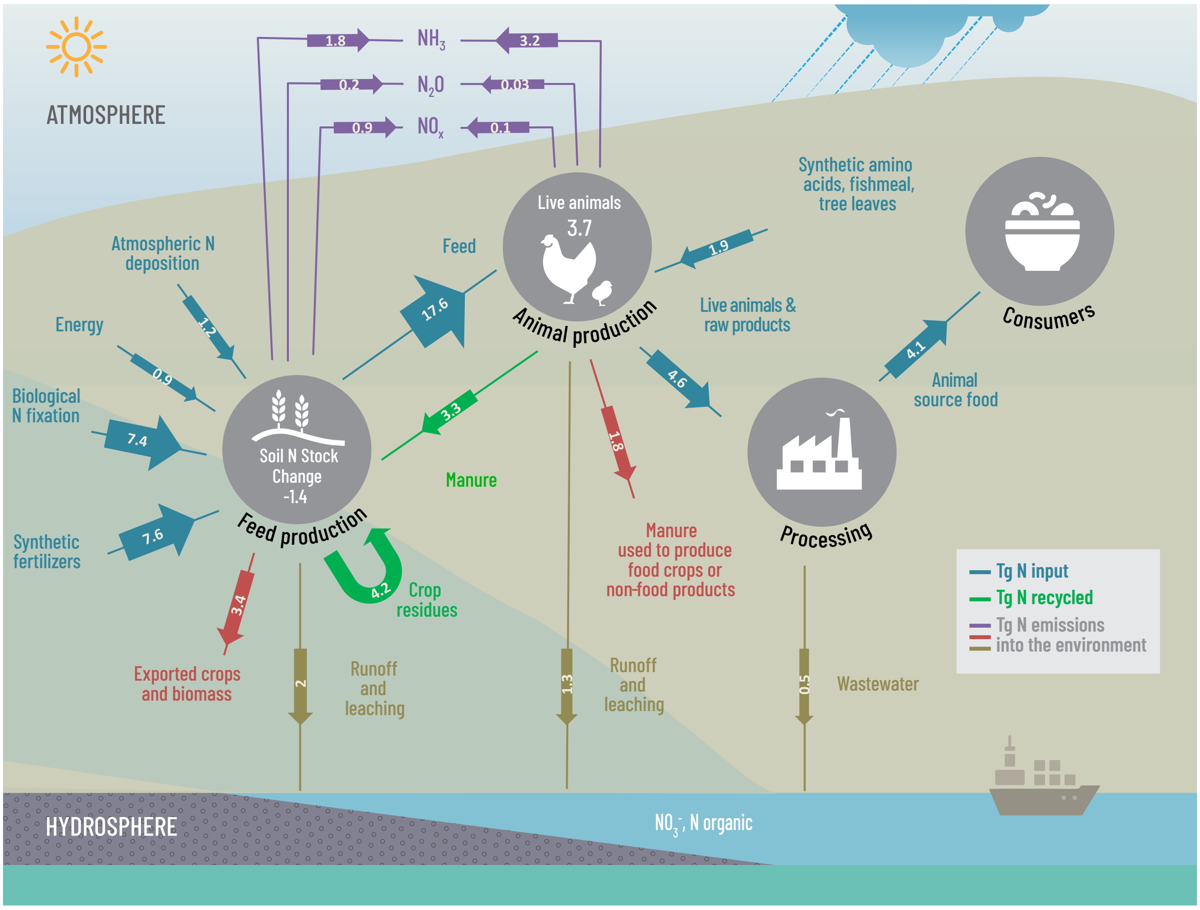
FIGURE A.3

**Global N flows and sources of N compounds emissions allocated to pigs for 2010 (Tg N/yr)**




*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F. *et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

FIGURE A.4
**Global N flows and sources of N compounds emissions allocated to poultry for 2010 (Tg N/yr)**




*Source:* Authors' elaboration based on **Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F.** ***et al.*** 2020. Nitrogen emissions along global livestock supply chains. *Nature Food*, 1(7): 437–446. https://doi.org/10.1038/s43016-020-0113-y

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