Nitrogen

Review

Climate Change and Nitrogen Dynamics: Challenges and Strategies for a Sustainable Future

Aline Viancelli and William Michelon *

Universidade do Contestado, Victor Sopelsa, 3000, Concórdia 89711-330, SC, Brazil; alineviancelli@unc.br
* Correspondence: william@unc.br; Tel.: +55-49-3441-1000

Abstract: Global warming driven by climate change has profound impacts on nitrogen dynamics in terrestrial and aquatic ecosystems. The increased emissions of greenhouse gases alter the distribution and availability of nitrogen, which is a critical nutrient for all living organisms. This review examines the connections between climate change and nitrogen cycling, highlighting the adverse effects on ecosystem health and productivity. The proliferation of nitrogen pollution due to agricultural runoff, industrial effluents, and urban wastewater aggravates eutrophication, leading to significant environmental and economic consequences. The imbalance in nitrogen availability not only affects plant growth and soil fertility but also disrupts aquatic ecosystems, resulting in harmful algal blooms and hypoxic conditions. Effective mitigation and adaptation strategies are essential to addressing these challenges. Sustainable agricultural practices, such as precision farming and the use of slow-release fertilizers, along with robust policies and innovative technologies, like biochar application and nitrification inhibitors, are essential in managing nitrogen levels. This review underscores the importance of interdisciplinary approaches that involve integrating insights from ecology, agronomy, and the social sciences to develop comprehensive solutions. Future research should focus on long-term studies to assess the cumulative impacts of climatic changes on nitrogen availability and ecosystem health to guide policies and management practices for sustainable development.

Keywords: nitrogen cycling; eutrophication; nitrogen pollution; ecosystem health

check for updates

Citation: Viancelli, A.; Michelon, W. Climate Change and Nitrogen Dynamics: Challenges and Strategies for a Sustainable Future. *Nitrogen* **2024**, *5*, 688–701. https://doi.org/10.3390/nitrogen5030045

Academic Editor: Marouane Baslam

Received: 9 July 2024
Revised: 30 July 2024
Accepted: 6 August 2024
Published: 8 August 2024

1. Introduction

Global warming, which is driven by climate change as a consequence of greenhouse gas emissions from anthropogenic activities, has caused significant alterations in terrestrial and aquatic ecosystems [1,2]. The environmental stress, such as hotter, drier, and cooler areas, as well as increased frequencies of droughts and floods, is causing changes in all living organisms and altering the distribution of inorganic and organic compounds [3]. These compounds are important to plant growth, especially those plants used as food sources for humans and animals. One of these important compounds is nitrogen.

Nitrogen is important to all living organisms since it is part of proteins, deoxyribonucleic acid (DNA), carbohydrates, fibers, and chlorophyll [4]. Nitrogen gas (N_2) is the major compound (78%) of the atmospheric air [5], and enters into organism systems through N deposition and/or fixation performed by plants and microbes. The deposition process is the direct input of ammonia (NH_3), nitric oxide (NO), nitrite (NO_2), nitric acid (HNO_3), and nitrates (NO_3) from the atmosphere into organisms, for example, by being carried from the atmosphere to the soil by precipitation [6]. Nitrogen fixation is the conversion of N_2 into NH_3, NO_2, and NO_3, such as in legume plants, where N_2 is fixed in the nodule by the symbiosis of plant and bacteria [6].

Nature and its biodiversity rely on the presence, even if limited, of reactive nitrogen (Nr) species within the ecosystem. Nr includes all nitrogen compounds except N_2, such as NH_3, NO_3, NOx, nitrous oxide (N_2O), amines, and organic nitrogen forms. Naturally occurring Nr is mainly generated through processes like lightning, wildfires, and biological nitrogen fixation. Human innovation has led to the creation of synthetic Nr fertilizers, which are employed to enhance agricultural productivity and meat production [7,8]. China, India, and the USA contribute significant portions of the global Nr emissions, with 20%, 11%, and 10%, respectively. Together with Brazil, which accounts for 6.1% of global Nr emissions, these four populous countries represent 47% of the total global Nr emissions [9].

The global production of nitrogen has significantly increased over recent decades to meet the demands for food from the growing population. However, this increased production has led to a rise in nitrogen pollution due to excessive use and poor management practices. Excess nitrogen not only contributes to air and water pollution but also impacts soil health and biodiversity [10]. Among these changes, nitrogen availability in the environment has emerged as an important factor influencing the ecosystem health, especially N_2O, which is a greenhouse gas with 298 times the warming potential of carbon dioxide (CO_2) [11].

The global nitrogen cycle features complex connections between biological, chemical, and physical processes that regulate the availability of nitrogen in various essential forms for all living creatures. Nitrogen, which is an essential nutrient for plant and aquatic organisms that frequently limits growth and yield, enters ecosystems, where it is recycled through the plant–soil–microbe system [12]. However, nitrogen is increasingly subject to imbalances due to climatic variations [13]. Climate oscillations in temperature, for example, change the soil microbial community physiology, which accelerates the rates of biological processes, such as nitrification and denitrification, and thus, alters the forms and availability of nitrogen in ecosystems. This is especially evident in agricultural land, where soil release is responsible for 56–70% of all atmospheric N_2O emissions [11].

Changes in precipitation patterns intensify the nitrogen runoff from agricultural areas into water bodies, which increases water pollution [12]. This process can lead to excessive algal growth, which, in turn, affects aquatic life by reducing the available oxygen levels [14]. Moreover, imbalances in nitrogen availability can alter the food web dynamics and aquatic biodiversity [15].

Understanding the relationship between global warming and nitrogen availability is important for developing mitigation and adaptation strategies to address climate change. This review explores the mechanisms through which global warming influences nitrogen cycling in the environment, the resulting impacts on the water quality and ecosystem health, and potential solutions to mitigate these adverse effects. Key mitigation strategies include improving the nitrogen use efficiency in agriculture, adopting sustainable farming practices, and enhancing policies and regulations to effectively manage nitrogen emissions.

2. Production and Consumption of Fertilizers Worldwide

The global consumption of nitrogen fertilizers surged from 18.6 million metric tons in 1965 to 119.2 million tons in 2021, which marked an increase of approximately 540.86% (Figure 1). Asia led this increase, with East and South Asia consuming the most. In 2019, China used about 23 million metric tons of nitrogenous fertilizers [16]. This rise reflects the growing demand for higher agricultural productivity and can cause significant implications for nitrogen cycling and environmental health.

In the early 2000s, the nitrogen fertilizer production was about 95 million metric tons. This increased to nearly 123 million tons by 2019, before slightly declining to 118.55 million tons in 2021 (Figure 2). This trend underscores the importance of nitrogen in sustaining global food production but also highlights the environmental challenges associated with its use [16].

The most widely consumed nitrogenous fertilizer is urea, with its high nitrogen content and relatively low cost (Figure 3). In 2021, urea consumption reached 53.83 million metric tons of nitrogen. Complex NPK (nitrogen, phosphorus, and potassium) fertilizers were the second most consumed, with approximately 17.31 million tons of nitrogen used. Ammonium phosphate, which is more widely used than ammonium nitrate, is valued for its effectiveness in providing essential nutrients for plant growth and its suitability for various soil types and conditions [16,17]. Ammonium nitrate is also significant due to its high effectiveness in different soils and climates [16].

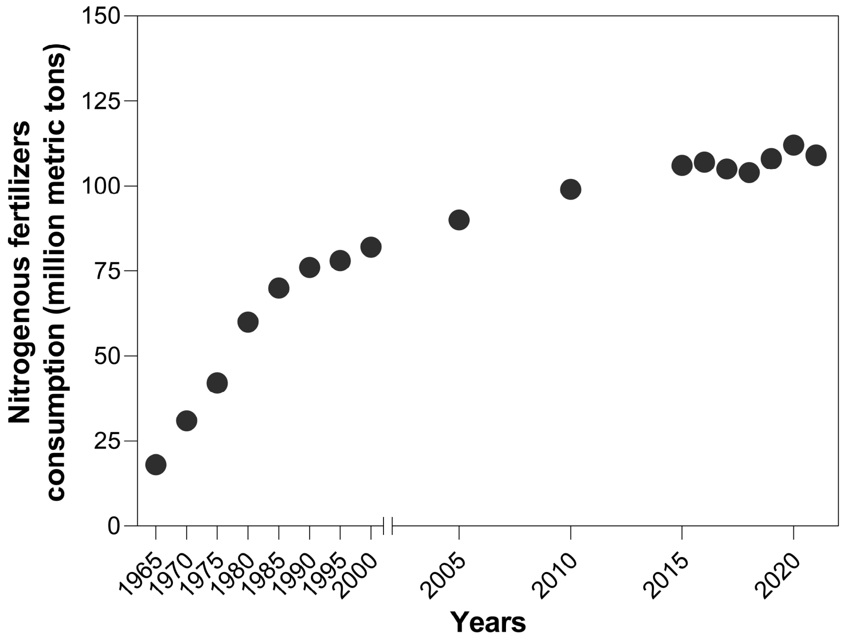

Figure 1. Global consumption of nitrogenous fertilizers over time.

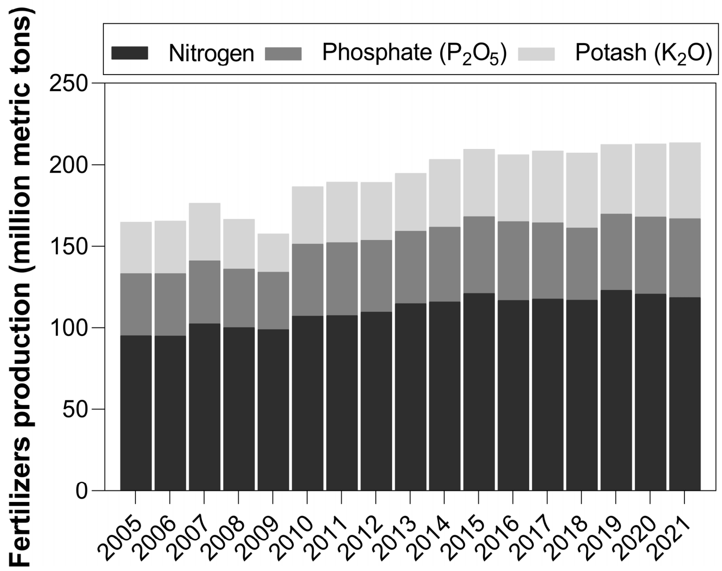

Figure 2. Global fertilizer production over time.

Although N is essential for plant growth, it often exceeds the capacity of crops to absorb it, leading to excess N leaching into rivers, lakes, and coastal waters, and consequently, causing environmental degradation. Moreover, the production of nitrogen fertilizers is energy intensive and contributes to greenhouse gas emissions, which further exacerbates climate change [18–20].

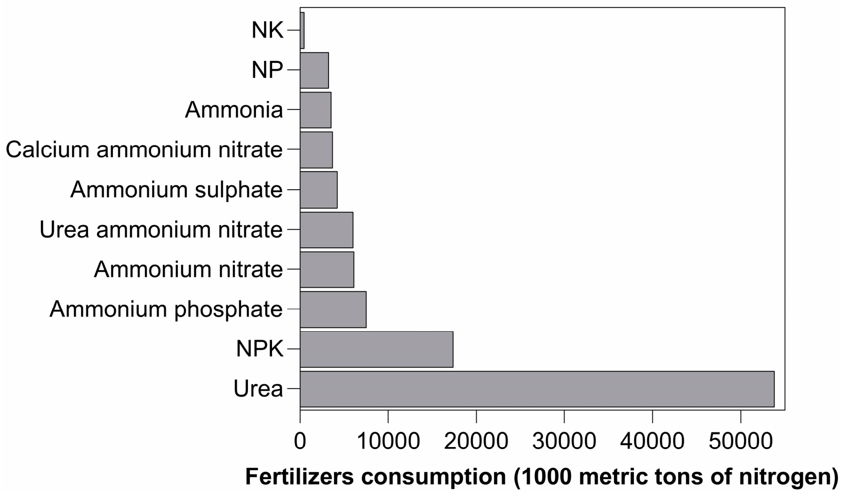

Figure 3. Global consumption of different nitrogen fertilizers in 2021.

3. Nitrogen Pollution Sources

Understanding the sources of nitrogen pollution is essential for developing effective mitigation strategies. Major sources include agricultural runoff, industrial effluents, and domestic wastewater [21,22].

Since the early 1970s, the widespread pollution of water bodies by NO_3-N due to agricultural intensification in industrialized countries of North America and Western and Central Europe has been a major concern [19]. This pollution directly resulted from the extensive application of nitrogen fertilizers in these regions. Globally, 60% of areas with elevated NO_3-N levels in groundwater are found in croplands [23]. Although only 20% of the total cultivated land is under irrigated agriculture, this land accounts for about 40% of global food production. Consequently, the use of nitrogen fertilizers and the subsequent loss of NO_3-N to natural water bodies are significantly higher in irrigated croplands than in rain-fed agriculture [19].

Domestic and industrial wastewaters are two major sources of nitrogen pollution in urban rivers [24]. Human excreta, which is the primary component of domestic wastewater, typically maintains stable concentration ratios of nitrogen, carbon, and phosphorus [25]. A 72-month monitoring dataset of water quality was used to assess the role of urban rivers in regional nitrogen pollution and their response to changes in human activities. The concentrations of ammonium nitrogen in urban rivers were found to be 3–5 times higher than those in regional rivers, indicating that urban rivers have become significant reservoirs of reactive nitrogen and hotspots of regional pollution [26].

Kaushal et al. [27] investigated the fate and transport of nonpoint N in forested, agricultural, and urbanized watersheds using combined watershed N mass balances. They found that the annual N retention was 55% in agricultural watersheds, 68% in urbanized watersheds, and 82% in forest watersheds. Their analysis revealed that wastewater was a significant NO_3 source in urbanized streams during baseflow. The correlation patterns indicated a mix between atmospheric deposition and wastewater in urban watersheds, with atmospheric sources contributing approximately 50% at peak storm N loads. Furthermore, denitrification processes appeared effective at removing septic-system- and agriculturally derived N, although N from leaking sewers was less susceptible to denitrification [27]. Overall, nonpoint sources, such as atmospheric deposition, wastewater, and fertilizer, showed varying levels of susceptibility to watershed N export. Significant changes in NO_3 sources due to runoff underscore the necessity of anticipating source changes in response to climate and storm events for the effective management of nonpoint N pollution [27].

Lee et al. [28] utilized a global land biosphere model to analyze historical terrestrial–freshwater nitrogen budgets by considering the impacts of anthropogenic N inputs, atmospheric CO_2, land use, and climate change. Their estimates indicate that globally, land sequesters approximately 11% of the annual N inputs. However, some river basins sequester more than 50% of their N inputs, thereby buffering coastal waters against eutrophication and mitigating greenhouse gas-induced warming. Conversely, other basins, particularly in the tropics, release more than 25% of their N inputs. This discrepancy is largely due to recent deforestation, agricultural intensification, and the export of land N storage, which create significant N pollution sources [28]. The tropics, despite covering only 34% of the global land area and receiving much lower amounts of fertilizers compared with the extratropics, are responsible for producing 56 ± 6% of the global land N pollution. Consequently, tropical land use must be carefully considered in global N pollution management strategies [28].

Excessive nitrogen can lead to nutrient imbalances, while deficiencies can limit plant growth. Efficient nitrogen management is essential to ensure sustainable agricultural practices and food security [29]. The efficient use of NPK fertilizers is necessary for food security and environmental preservation. Nutrient imbalances, particularly the disrupted N ratio from anthropogenic activities, like crop fertilization and the expansion of N-fixing crops, have continuously increased the soil N ratios, which has affected terrestrial and aquatic ecosystems by altering the species composition and functionality, and thus, threatening global biodiversity [30].

4. Impacts on Terrestrial Ecosystems

In terrestrial ecosystems, nitrogen is a limiting nutrient that regulates primary productivity [31]. The availability of nitrogen directly impacts plant growth, soil fertility, and ecosystem dynamics. Climate change, through mechanisms such as increased temperatures and altered precipitation patterns, influences the nitrogen cycle and its availability in the soil [32]. In terrestrial environments, both nitrogen deficiency and excess can interfere with the lives of organisms.

In forest ecosystems, the nitrogen cycle is influenced by the vegetation type, as the cover structure and root distribution affects the soil temperature, substrate availability, litter composition, microbial diversity, nitrogen leaching, and gaseous nitrogen loss [33]. In a forest, organisms compete for the limited nitrogen released through depolymerization and mineralization processes. These organisms involve mature trees, naturally regenerating plants, various woody and herbaceous understory species, mycorrhizal fungi, and free-living fungi and bacteria, both near and far from the rhizosphere. Direct competition between these organisms can be mitigated by spatial and temporal separation in N acquisition. This includes occupying distinct soil compartments and/or preferentially absorbing N during different seasons, as well as having varying preferences for N sources, such as inorganic versus organic N compounds [34–36].

Tropical forests are important in the N cycle since they contribute about 70% of terrestrial N fixation and release about 50% of N_2O [37], and thus, they are the important regulators of biogeochemical cycles. However, these environments are experiencing high pressure from deforestation and rapid rates of land conversion to agriculture, which influence the nitrogen cycle [38,39].

In a tropical forest, the continuous addition of nitrogen compounds from the atmosphere, primarily due to human activities, such as industrial emissions and agricultural practices, can lead to a condition known as nitrogen saturation, where the supply of nitrogen exceeds the biological demand. In forest ecosystems, this excess nitrogen results in increased soil acidification, which negatively impacts tree growth and carbon storage and shifts the nutrient limitation from nitrogen to phosphorus. Additionally, the increased acidity and altered nutrient balance can reduce the plant diversity [40].

The historical and current decreases in N availability in the forest soil may have been caused by altered ecosystem disturbance regimes. N has been continuously—and is becoming more so—exported from ecosystems and transferred to the most populous watersheds through the harvesting of biomass (wood, vegetables, meat). Many places are experiencing an increase in the frequency of fires, which is linked to increased N losses across decadal time periods [40,41]. It was discovered that regular burning reduces soil nitrogen by over 40% over the course of six decades in savanna grasslands and broadleaf forests [42]. In cases where supplemental feeding is not practical (e.g., most rangelands that are grazed without the use of fertilizers) and when frequent N losses occur without significant inputs, a long-term decline in N availability will be difficult to avoid [43].

Soil acidification occurs when ammonium-based fertilizers are converted to NO_3, releasing hydrogen ions in the process. This increased acidity can leach essential nutrients, like calcium and magnesium, from the soil, which further reduces its fertility [44]. High nitrogen levels can also result in the loss of biodiversity. Certain plant species, particularly those adapted to low-nitrogen conditions, may be outcompeted by nitrogen-loving species, which leads to a decline in species richness and diversity. This shift can have cascading effects on the entire ecosystem, including reduced habitat complexity, which affects herbivores and higher trophic levels [45], and diminished ecosystem resilience [46–48].

Dal Molin et al. [49] evaluated the N release rate and soil acidification caused by N fertilizers. Treatments included urea, ammonium nitrate, and six coated fertilizers, with all of them applied at 300 mg kg^{-1} N, along with a control (without N). Fertilizers were mixed with 100 g of soil in plastic pots. Mineral N and soil pH were measured at 4, 7, 14, 21, 28, 35, 42, 49, and 56 days post-application. All fertilizers, except ammonium nitrate, decreased the soil pH over time. Ammonium was the dominant N form in the soil up to day 35 for all fertilizers, except ammonium nitrate. Sulfur + polymer-coated urea released 90% of its N from day 42, unlike other fertilizers. Coated N fertilizers did not effectively reduce the soil acidification, with only sulfur + polymer-coated urea showing slow-release characteristics [49].

While many experiments demonstrated that random species losses affect ecosystem functionality, human-induced biodiversity losses are rarely random. Using data from long-term grassland field experiments, Isbell et al. [50] tested the direct effects of chronic nutrient enrichment on ecosystem productivity and the indirect effects mediated by species losses. The findings revealed that the ecosystem productivity declined the most significantly in the plots that experienced the highest species loss. Chronic nitrogen addition resulted in the nonrandom loss of initially dominant native perennial C4 grasses. This selective loss led to a productivity decline that was twice as great as that observed with random species loss in a nearby biodiversity experiment. Thus, while chronic nitrogen enrichment initially boosted the productivity, it also caused the loss of dominant plant species, which resulted in substantial diminishing returns from nitrogen fertilization. In contrast, elevated CO_2 did not reduce the grassland plant diversity and consistently enhanced the productivity over time. These results support the hypothesis that the long-term impacts of anthropogenic environmental changes on ecosystem functioning significantly depend on how these changes reduce biodiversity and restructure communities [50].

High nitrogen levels can lead to the proliferation of nitrogen-loving species, often at the expense of other plant species [51]. For instance, fast-growing grasses may outcompete slower-growing forbs, which leads to homogenized plant communities with lower species richness. Such changes can reduce the resilience of ecosystems to environmental stressors, which makes them more susceptible to disturbances, such as drought, pests, and diseases [52].

These alterations are the consequences of the temperature on microbial activity in the soil, which influences organic matter decomposition, and consequently, nitrogen release. The interactions between temperature, soil moisture, and microbial activity are complex and can vary significantly across different ecosystems, which complicates the prediction of the exact effects of climate change on nitrogen cycling [53].

A comprehensive understanding of nitrogen cycling also requires examining the impacts of increased atmospheric CO_2 concentrations. Elevated CO_2 levels can enhance plant growth and nitrogen uptake, which potentially leads to greater nitrogen sequestration in plant biomass [54].

5. Impacts on Aquatic Ecosystems

In aquatic ecosystems, nitrogen pollution often leads to eutrophication, which is a process characterized by excessive nutrient enrichment that stimulates algal blooms. These blooms can have severe ecological and economic consequences. Eutrophication is a major environmental issue in aquatic ecosystems that is often driven by agricultural runoff containing high levels of nitrogen and phosphorus. The resultant algal blooms and later algal death can lead to the decomposition of this algal biomass by aerobic bacteria, which consequently depletes the oxygen levels in the water, and thus, creates hypoxic or anoxic conditions, commonly referred to as "dead zones", where most aquatic life cannot survive [55].

Certain types of algal blooms, known as harmful algal blooms (HABs), produce toxins that are detrimental to both aquatic life and human health. These toxins can accumulate in shellfish, which pose risks to human consumers, and can cause large-scale deaths of fish and other marine organisms [56]. Nitrogen pollution leads to the degradation of water quality, which affects its suitability for drinking, recreational activities, and supporting biodiversity [57].

The presence of high NO_3 levels in drinking water sources has become an increasing concern due to their negative impacts on human health. The maximum contaminant level for NO_3 in public drinking water is 10 mg/L as NO_3-N, which is equivalent to the World Health Organization (WHO) guideline of 50 mg/L as NO_3 (11.3 mg/L NO_3-N) [58]. When water with nitrate concentrations above 10 mg/L [59] is ingested in high quantities, NO_3 can be converted into NO_2 in the human body, which can interfere with the blood's ability to transport oxygen. This process is particularly dangerous for infants, which leads to a condition known as methemoglobinemia, or "blue baby syndrome". In this condition, hemoglobin is oxidized to a form that cannot effectively carry oxygen, which causes cyanosis and, in severe cases, hypoxia [59]. An awareness of contamination sources and the implementation of sustainable agricultural practices are essential for protecting drinking water sources and preventing the adverse health effects associated with NO_3 [60].

The economic impacts of nitrogen pollution in aquatic ecosystems are significant due to losses in commercial and recreational fisheries, reduced tourism revenue due to degraded water bodies, and increased costs for water treatment [61]. The Gulf of Mexico, for instance, experiences a large dead zone each summer due to nutrient runoff from the Mississippi River, which highlights the far-reaching impacts of nitrogen pollution exacerbated by climate change [62].

6. Mitigation and Adaptation to Climate Change

Addressing the impacts of climate change on nitrogen availability requires a combination of mitigation and adaptation strategies. Implementing sustainable agricultural practices, such as precision farming, cover cropping, and the use of slow-release fertilizers, is important to reduce nitrogen runoff and enhance the nitrogen use efficiency [63]. Restoring and preserving natural wetland areas is also effective since wetlands act as filters that trap and remove nutrients from runoff before they reach open water [64,65].

Strong policies and regulations are essential to control nitrogen emissions from industrial and agricultural sources. This includes setting limits on fertilizer application, improving wastewater treatment standards, and promoting best management practices [66]. Policies that incentivize sustainable practices and support research and innovation are critical for developing new technologies and practices for managing nitrogen in the environment. Advancements in soil testing, crop breeding for improved nitrogen uptake, and real-time monitoring systems for nutrient levels are key areas of focus [67,68]. Increasing public awareness and education about the impacts of nitrogen pollution and climate change can drive behavioral changes and give support for environmental policies. Educational campaigns and community engagement initiatives can promote sustainable practices at the local level, fostering a culture of environmental management [68].

Technical alternatives have also been reported. Biochar, which is a form of charcoal produced from organic materials, has shown promise in reducing greenhouse gas emissions, specifically N_2O, from soils. Both initial biochar application and subsequent reapplications have been effective in reducing N_2O emission peaks during seasonal events, leading to a 1–34% decrease in cumulative N_2O emissions over the study period compared with treatments without biochar [69]. This reduction is primarily attributed to the increase in soil pH due to biochar application, rather than the reduction in NH_4^+ and NO_3^- levels. These findings suggest that biochar application, either as a single treatment or with reapplications, with or without nitrogen fertilizer, is a viable strategy for enhancing the sustainability of intensive agriculture. The study recommended applying biochar at a rate of 20 t ha^{-1} under conventional nitrogen application, with a second application of the same amount after four years. For more intensive fertilization, the recommended biochar input is also 20 t ha^{-1}, but without the need for reapplication [69].

The application of biochar and urease inhibitors in urea-fertilized soils can significantly reduce NH_3 emissions and N_2O while simultaneously enhancing the nitrogen use efficiency and crop productivity in wheat [70]. Specifically, the combined use of biochar and urease inhibitors reduced the NH_3 emissions by 69% and N_2O emissions by 53% while increasing the biomass by 38% and grain yield by 22% compared with urea application alone. The introduction of biochar also positively influenced the nitrogen dynamics in the soil, which promoted greater retention of NH_{4+} and reduced NO_3^- concentrations, and thereby mitigated nitrogen losses through volatilization and leaching [70].

The use of urease inhibitors in urea-based fertilizers offers significant benefits by reducing nitrogen losses due to volatilization, thereby improving the nitrogen use efficiency and potentially lowering overall fertilizer costs [71]. Recent experiences highlight that these inhibitors can enhance the crop yield and quality by ensuring a more stable nitrogen supply. Given the recent spike in fertilizer prices, the cost-effectiveness of urease inhibitors becomes particularly relevant, as they can reduce the need for frequent fertilizer applications [71,72]. The application of urease inhibitors, such as N-(n-butyl) thiophosphoric triamide and N-(propyl) thiophosphoric triamide, in various crop types, including rice, barley, cotton, and maize, demonstrated significant benefits in enhancing the crop growth and nitrogen use efficiency. For instance, rice and maize showed improved grain yields and nitrogen uptake, while cotton exhibited an enhanced leaf chlorophyll content and fiber quality [73]. Additionally, the use of these inhibitors reduced the ammonia volatilization and nitrogen losses, particularly under conditions conducive to volatilization or denitrification, as observed in wheat and pasture systems [73]. However, the effectiveness of urease inhibitors varied depending on the crop type, soil conditions, and environmental factors, highlighting the need for tailored applications to optimize nitrogen management in agricultural practices [73].

Nitrification inhibitors can be used to reduce N_2O emissions from agricultural soils, but their effectiveness can vary significantly depending on the soil temperature and moisture levels. For example, Guo et al. [74] studied the efficacy of nitrification inhibitors in mitigating N_2O emissions under various soil temperature and moisture conditions. They found that 3,4-dimethylpyrazole phosphate and 3-methylpyrazole were more effective than dicyandiamide in sandy soil at different moisture levels and temperatures. Specifically, 3,4-dimethylpyrazole phosphate was the most effective by completely inhibiting N_2O emissions at temperatures between 15–25 °C and up to 80% water-holding capacity. As a result, this inhibitor is recommended for use in agricultural lands [74]. In a study on the use of 3,5-dimethylpyrazole as a nitrification inhibitor in soils, Lu et al. [75] found that 3,5-dimethylpyrazole application significantly inhibited nitrification, which reduced the soil nitrate reductase activity while increasing the nitrite reductase and dehydrogenase activities. The inhibitory effect was dose dependent, with the concentration of 0.025 g/kg dry soil showing the strongest inhibitory effect on nitrogen transformation. These findings suggest that 3,5-dimethylpyrazole can be an effective tool for mitigating nitrate leaching and greenhouse gas emissions by promoting a slower nitrogen transformation in the soil, which can be beneficial for low-carbon agriculture and environmental sustainability [75].

Ren et al. [76] observed that the use of UAN (urea–ammonium nitrate) fertilizer significantly reduced N_2O emissions and enhanced the nitrogen use efficiency compared with conventional urea. Specifically, UAN reduced the N_2O emissions by 39.3% and improved the nitrogen recovery efficiency by 31.2%. Additionally, UAN increased the maize grain yield by 9.1% compared with urea, indicating a more efficient nitrogen uptake and utilization. The study also found that UAN led to lower soil NO_3-N and NH_4-N concentrations in the surface soil layer, which reduced the risk of nitrogen leaching and environmental pollution. Dattamudi et al. [77] demonstrated that the use of urea as a nitrogen source results in significantly higher N_2O emissions compared with UAN in sugarcane production. The study found that the N_2O emissions from the urea-treated plots were 1.43 to 3.16 times greater than those from the UAN-treated plots. Specifically, the emission factors for N_2O were 3.52% and 4.45% for urea under residue-burned and residue-retained management practices, respectively, compared with 1.67% and 2.46% for UAN under the same conditions. The majority of these emissions occurred within four weeks following the nitrogen application, where the residue-retained treatment showed higher emissions due to the increased soil moisture content [77].

The study by Ren et al. [78] investigated the effects of combining UAN with urease and nitrification inhibitors on the yield, nitrogen efficiency, and ecological impacts in summer maize under fertigation. The treatments included no inhibitor, nitrification inhibitor, urease inhibitor, and a combination of both inhibitors. The results indicated that integrating UAN with either inhibitor reduced the cumulative N_2O emissions, global warming potential, and greenhouse gas intensity. Specifically, the cumulative N_2O emissions were decreased by 45.7% with the inhibitor combination, 25.3% with the urease inhibitor, and 35.8% with the nitrification inhibitor treatments compared with the no inhibitor treatment [78]. The application of urease inhibitors significantly reduced the NH_3 volatilization, where the urease inhibitor and inhibitor combination treatments showed reductions of 30.7% and 23.4%, respectively, while the nitrification inhibitor treatment increased it by 12.3%. Furthermore, the use of inhibitors with UAN significantly enhanced the NO_3-N and NH_4-N concentrations in the 0–20 cm soil layer and minimized the NO_3-N leaching into deeper soil layers [78].

Nikolajsen et al. [79] examined the effects of different nitrogen fertilizers on ammonia emissions and crop yield. The research found that when UAN was used without any inhibitors, it led to the highest ammonia emissions, with up to 17% of the applied nitrogen being lost as NH_3. In contrast, when the urease inhibitor N-(n-butyl) thiophosphoric triamide was used, the ammonia emissions were reduced to 11% [79]. Furthermore, the combination of NBPT with the nitrification inhibitor 3,4-Dimethylpyrazole succinic acid lowered emissions even further to 7%. The study, which was conducted on *Triticum aestivum* L. (winter wheat) grown in a cool temperate climate with sandy loam soil, also observed a wheat grain yield increase of up to 14% with the use of these inhibitors. These results underscore the effectiveness of inhibitors in reducing nitrogen losses and enhancing crop yield, which are particularly important under the varying temperature and precipitation conditions influenced by climate change [79].

A study conducted by Peng et al. [80] found that applying *Streptomyces* JD211 to rice soils significantly reduced the N_2O emissions by 32.0–68.6% across treatment groups with 0.1–1.0% concentrations. *Streptomyces* JD211 maintains soil mineral nitrogen content during rice growth by lowering NH_4^+-N and NO_3—N levels in the early stages of cultivation. It also alters the rice rhizosphere microbial community by increasing beneficial bacteria, like *Burkholderia* and *Streptomyces*, while reducing NO_2-oxidizing bacteria, such as *Nitrospirota*. Furthermore, *Streptomyces* JD211 impacts nitrogen cycle functional genes by decreasing those involved in N_2O production (hao, norB, norC) and increasing those related to N_2O reduction (nosZ, hcp). Additionally, the agent promotes rice seedling growth by enhancing the seedling height and root number and length, indicating its potential to improve the crop yield and soil health, which makes it a promising tool for sustainable agriculture [80].

This holistic approach, which integrates sustainable agricultural practices, robust policies, innovative research, and public education, is essential for effectively managing nitrogen availability and mitigating its environmental impacts. By addressing the issue from multiple angles, we can develop comprehensive solutions that protect our environment, ensure food security, and promote the health and well-being of future generations. The combined efforts of policymakers, researchers, farmers, and the public will be necessary in achieving these goals and creating a sustainable future for all.

7. Conclusions and Perspectives

Climate change is inherently linked to nitrogen availability in terrestrial and aquatic ecosystems. Global warming and changes in precipitation patterns affect nitrogen cycling, which results in imbalances that can have negative consequences for the ecosystem health. Developing and implementing mitigation and adaptation strategies is essential to addressing these challenges and ensuring the sustainability of ecosystems. Continuous investments in research and innovation are fundamental to improving our understanding and management of nitrogen availability in a changing world.

Future research should prioritize understanding the interactions between climate change and nitrogen cycling across various ecosystems. Long-term studies are essential to assess the cumulative impacts of climatic changes on nitrogen availability and ecosystem health. These studies can provide valuable insights into how different ecosystems respond to changes in temperature, precipitation, and CO_2 levels, thereby identifying vulnerabilities and informing adaptation strategies.

Developing more accurate predictive models to anticipate the effects of climate change on nitrogen cycling is also important. These models can guide policies and management practices that minimize negative environmental impacts and promote the sustainability of natural resources.

Additionally, research should explore the socio-economic impacts of nitrogen imbalances, especially in regions that are heavily dependent on agriculture and fisheries. Understanding how changes in nitrogen availability affect food security, livelihoods, and economic stability can help to design more effective and equitable mitigation and adaptation strategies. Conducting case studies and regional assessments will provide detailed insights into the specific challenges and solutions related to nitrogen cycling in different geographic areas, and thus, highlight best practices and inform local and regional policy decision.

Innovative technologies, such as biochar application, nitrification inhibitors, and advanced wastewater treatment processes, offer promising solutions for managing nitrogen in the environment. Biochar can enhance soil fertility and increase nitrogen retention, which reduces the need for fertilizers and mitigating nitrogen runoff. Nitrification inhibitors can slow the conversion of ammonium to nitrate, which reduces nitrogen losses through leaching and denitrification. Advanced wastewater treatment processes can improve nitrogen removal from domestic and industrial effluents, which reduces their impact on the water quality. Evaluating these technologies for effectiveness, cost, and scalability is essential to determine their feasibility and potential for widespread adoption.

Interdisciplinary approaches are critical for addressing the complex issues of nitrogen cycling and climate change. Integrating insights from ecology, agronomy, economics, and social sciences can provide a more comprehensive understanding of the interactions between climate change and nitrogen dynamics. Collaborative research efforts that bring together experts from different fields can lead to more effective solutions and inform policies and practices that promote sustainability.

Author Contributions: Conceptualization, A.V. and W.M.; investigation, A.V. and W.M.; writing—original draft preparation, A.V. and W.M.; writing—review and editing, A.V. and W.M. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding.

Data Availability Statement: The datasets generated for this study are available upon request to the corresponding author.

Conflicts of Interest: The authors declare no conflicts of interest.

References

1. Priya, A.K.; Muruganandam, M.; Rajamanickam, S.; Sivarethinamohan, S.; Gaddam, M.K.R.; Velusamy, P.; Gomathi, G.; Ravindiran, G.; Gurugubelli, T.R.; Muniasamy, S.K. Impact of Climate Change and Anthropogenic Activities on Aquatic Ecosystem—A Review. *Environ. Res.* **2023**, *238*, 117233. [CrossRef]
2. Smith, P.; Martino, D.; Cai, Z.; Gwary, D.; Janzen, H.; Kumar, P.; McCarl, B.; Ogle, S.; O'Mara, F.; Rice, C.; et al. Greenhouse Gas Mitigation in Agriculture. *Philos. Trans. R. Soc. B Biol. Sci.* **2008**, *363*, 789–813. [CrossRef] [PubMed]
3. Mandal, D.; Roy, T. Climate Change Impact on Soil Erosion and Land Degradation; 2024; pp. 139–161.
4. Daniel-Vedele, F.; Krapp, A.; Kaiser, W.M. *Cellular Biology of Nitrogen Metabolism and Signaling*; Springer: Berlin/Heidelberg, Germany, 2010; pp. 145–172.
5. Mosier, A. Exchange of Gaseous Nitrogen Compounds between Agricultural Systems and the Atmosphere. *Plant Soil* **2001**, *228*, 17–27. [CrossRef]
6. Edwards, T.M.; Puglis, H.J.; Kent, D.B.; Durán, J.L.; Bradshaw, L.M.; Farag, A.M. Ammonia and Aquatic Ecosystems—A Review of Global Sources, Biogeochemical Cycling, and Effects on Fish. *Sci. Total Environ.* **2024**, *907*, 167911. [CrossRef] [PubMed]
7. Schlesinger, W.H. On the Fate of Anthropogenic Nitrogen. *Proc. Natl. Acad. Sci. USA* **2009**, *106*, 203–208. [CrossRef]
8. Erisman, J.W.; Galloway, J.; Seitzinger, S.; Bleeker, A.; Butterbach-Bahl, K. Reactive Nitrogen in the Environment and Its Effect on Climate Change. *Curr. Opin. Environ. Sustain.* **2011**, *3*, 281–290. [CrossRef]
9. Oita, A.; Malik, A.; Kanemoto, K.; Geschke, A.; Nishijima, S.; Lenzen, M. Substantial Nitrogen Pollution Embedded in International Trade. *Nat. Geosci.* **2016**, *9*, 111–115. [CrossRef]
10. de Vries, W. Impacts of Nitrogen Emissions on Ecosystems and Human Health: A Mini Review. *Curr. Opin. Environ. Sci. Health* **2021**, *21*, 100249. [CrossRef]
11. Syakila, A.; Kroeze, C. The Global Nitrous Oxide Budget Revisited. *Greenh. Gas Meas. Manag.* **2011**, *1*, 17–26. [CrossRef]
12. Cleveland, C.C.; Houlton, B.Z.; Smith, W.K.; Marklein, A.R.; Reed, S.C.; Parton, W.; Del Grosso, S.J.; Running, S.W. Patterns of New versus Recycled Primary Production in the Terrestrial Biosphere. *Proc. Natl. Acad. Sci. USA* **2013**, *110*, 12733–12737. [CrossRef] [PubMed]
13. Jansson, J.K.; Hofmockel, K.S. Soil Microbiomes and Climate Change. *Nat. Rev. Microbiol.* **2020**, *18*, 35–46. [CrossRef]
14. Alhamarna, M.Z.; Tandyrak, R. Lakes Restoration Approaches. *Limnol. Rev.* **2021**, *21*, 105–118. [CrossRef]
15. Nessel, M.P.; Konnovitch, T.; Romero, G.Q.; González, A.L. Nitrogen and Phosphorus Enrichment Cause Declines in Invertebrate Populations: A Global Meta-analysis. *Biol. Rev.* **2021**, *96*, 2617–2637. [CrossRef] [PubMed]
16. Statista Consumption of Fertilizers Worldwide. Available online: https://www.statista.com/statistics/438967/fertilizer-consumption-globally-by-nutrient/ (accessed on 24 June 2024).
17. Margenot, A.J.; Lee, J. The Fate of Nitrogen of Ammonium Phosphate Fertilizers: A Blind Spot. *Agric. Environ. Lett.* **2023**, *8*, e20116. [CrossRef]
18. Dobermann, A.; Bruulsema, T.; Cakmak, I.; Gerard, B.; Majumdar, K.; McLaughlin, M.; Reidsma, P.; Vanlauwe, B.; Wollenberg, L.; Zhang, F.; et al. Responsible Plant Nutrition: A New Paradigm to Support Food System Transformation. *Glob. Food Sec.* **2022**, *33*, 100636. [CrossRef]
19. Bijay-Singh; Craswell, E. Fertilizers and Nitrate Pollution of Surface and Ground Water: An Increasingly Pervasive Global Problem. *SN Appl. Sci.* **2021**, *3*, 518. [CrossRef]
20. Aryal, B.; Gurung, R.; Camargo, A.F.; Fongaro, G.; Treichel, H.; Mainali, B.; Angove, M.J.; Ngo, H.H.; Guo, W.; Puadel, S.R. Nitrous Oxide Emission in Altered Nitrogen Cycle and Implications for Climate Change. *Environ. Pollut.* **2022**, *314*, 120272. [CrossRef] [PubMed]
21. Berger, M.; Canty, S.W.J.; Tuholske, C.; Halpern, B.S. Sources and Discharge of Nitrogen Pollution from Agriculture and Wastewater in the Mesoamerican Reef Region. *Ocean. Coast. Manag.* **2022**, *227*, 106269. [CrossRef]
22. Singh, B.J.; Chakraborty, A.; Sehgal, R. A Systematic Review of Industrial Wastewater Management: Evaluating Challenges and Enablers. *J. Environ. Manag.* **2023**, *348*, 119230. [CrossRef]
23. Shukla, S.; Saxena, A. Global Status of Nitrate Contamination in Groundwater: Its Occurrence, Health Impacts, and Mitigation Measures. In *Handbook of Environmental Materials Management*; Springer International Publishing: Cham, Switzerland, 2018; pp. 1–21.
24. Grimm, N.B.; Foster, D.; Groffman, P.; Grove, J.M.; Hopkinson, C.S.; Nadelhoffer, K.J.; Pataki, D.E.; Peters, D.P. The Changing Landscape: Ecosystem Responses to Urbanization and Pollution across Climatic and Societal Gradients. *Front. Ecol. Environ.* **2008**, *6*, 264–272. [CrossRef]
25. Ma, L.; Wang, F.; Zhang, W.; Ma, W.; Velthof, G.; Qin, W.; Oenema, O.; Zhang, F. Environmental Assessment of Management Options for Nutrient Flows in the Food Chain in China. *Environ. Sci. Technol.* **2013**, *47*, 7260–7268. [CrossRef]

26. Zhang, X.; Wu, Y.; Gu, B. Urban Rivers as Hotspots of Regional Nitrogen Pollution. *Environ. Pollut.* **2015**, *205*, 139–144. [CrossRef] [PubMed]
27. Kaushal, S.S.; Groffman, P.M.; Band, L.E.; Elliott, E.M.; Shields, C.A.; Kendall, C. Tracking Nonpoint Source Nitrogen Pollution in Human-Impacted Watersheds. *Environ. Sci. Technol.* **2011**, *45*, 8225–8232. [CrossRef]
28. Lee, M.; Shevliakova, E.; Stock, C.A.; Malyshev, S.; Milly, P.C.D. Prominence of the Tropics in the Recent Rise of Global Nitrogen Pollution. *Nat. Commun.* **2019**, *10*, 1437. [CrossRef]
29. Yadav, M.R.; Kumar, S.; Lal, M.K.; Kumar, D.; Kumar, R.; Yadav, R.K.; Kumar, S.; Nanda, G.; Singh, J.; Udawat, P.; et al. Mechanistic Understanding of Leakage and Consequences and Recent Technological Advances in Improving Nitrogen Use Efficiency in Cereals. *Agronomy* **2023**, *13*, 527. [CrossRef]
30. Penuelas, J.; Coello, F.; Sardans, J. A Better Use of Fertilizers Is Needed for Global Food Security and Environmental Sustainability. *Agric. Food Secur.* **2023**, *12*, 5. [CrossRef]
31. Knapp, A.K.; Carroll, C.J.W.; Fahey, T.J. Primary Production in Terrestrial Ecosystems: Patterns and Controls in a Changing World. In *Ecology and the Environment*; Springer: New York, NY, USA, 2014; pp. 1–36.
32. Yi, Q.; Curtright, A.J.; Horwath, W.R.; Zhu-Barker, X. Response of Soil Nitrogen Mineralization to Warming Temperatures Depends on Soil Management History. *Geoderma* **2023**, *440*, 116716. [CrossRef]
33. Rothe, A.; Huber, C.; Kreutzer, K.; Weis, W. Deposition and Soil Leaching in Stands of Norway Spruce and European Beech: Results from the Höglwald Research in Comparison with Other European Case Studies. *Plant Soil* **2002**, *240*, 33–45. [CrossRef]
34. Dannenmann, M.; Simon, J.; Gasche, R.; Holst, J.; Naumann, P.S.; Kögel-Knabner, I.; Knicker, H.; Mayer, H.; Schloter, M.; Pena, R.; et al. Tree Girdling Provides Insight on the Role of Labile Carbon in Nitrogen Partitioning between Soil Microorganisms and Adult European Beech. *Soil Biol. Biochem.* **2009**, *41*, 1622–1631. [CrossRef]
35. Schimel, J.P.; Bennett, J. Nitrogen Mineralization: Challenges of a Changing Paradigm. *Ecology* **2004**, *85*, 591–602. [CrossRef]
36. Rennenberg, H.; Dannenmann, M.; Gessler, A.; Kreuzwieser, J.; Simon, J.; Papen, H. Nitrogen Balance in Forest Soils: Nutritional Limitation of Plants under Climate Change Stresses. *Plant Biol.* **2009**, *11*, 4–23. [CrossRef] [PubMed]
37. Townsend, A.; Asner, G.; Cleveland, C. The Biogeochemical Heterogeneity of Tropical Forests. *Trends Ecol. Evol.* **2008**, *23*, 424–431. [CrossRef] [PubMed]
38. Hietz, P.; Turner, B.L.; Wanek, W.; Richter, A.; Nock, C.A.; Wright, S.J. Long-Term Change in the Nitrogen Cycle of Tropical Forests. *Science* **2011**, *334*, 664–666. [CrossRef]
39. Stork, N.E.; Coddington, J.A.; Colwell, R.K.; Chazdon, R.L.; Dick, C.W.; Peres, C.A.; Sloan, S.; Willis, K. Vulnerability and Resilience of Tropical Forest Species to Land-Use Change. *Conserv. Biol.* **2009**, *23*, 1438–1447. [CrossRef] [PubMed]
40. Pajares, S.; Bohannan, B.J.M. Ecology of Nitrogen Fixing, Nitrifying, and Denitrifying Microorganisms in Tropical Forest Soils. *Front. Microbiol.* **2016**, *7*, 01045. [CrossRef]
41. Pellegrini, A.F.A.; Ahlström, A.; Hobbie, S.E.; Reich, P.B.; Nieradzik, L.P.; Staver, A.C.; Scharenbroch, B.C.; Jumpponen, A.; Anderegg, W.R.L.; Randerson, J.T.; et al. Fire Frequency Drives Decadal Changes in Soil Carbon and Nitrogen and Ecosystem Productivity. *Nature* **2018**, *553*, 194–198. [CrossRef] [PubMed]
42. Abbott, B.W.; Rocha, A.V.; Shogren, A.; Zarnetske, J.P.; Iannucci, F.; Bowden, W.B.; Bratsman, S.P.; Patch, L.; Watts, R.; Fulweber, R.; et al. Tundra Wildfire Triggers Sustained Lateral Nutrient Loss in Alaskan Arctic. *Glob. Change Biol.* **2021**, *27*, 1408–1430. [CrossRef] [PubMed]
43. Mason, R.E.; Craine, J.M.; Lany, N.K.; Jonard, M.; Ollinger, S.V.; Groffman, P.M.; Fulweiler, R.W.; Angerer, J.; Read, Q.D.; Reich, P.B.; et al. Evidence, Causes, and Consequences of Declining Nitrogen Availability in Terrestrial Ecosystems. *Science* **2022**, *376*, eabh3767. [CrossRef]
44. Zhou, J.; Xia, F.; Liu, X.; He, Y.; Xu, J.; Brookes, P.C. Effects of Nitrogen Fertilizer on the Acidification of Two Typical Acid Soils in South China. *J. Soils Sediments* **2014**, *14*, 415–422. [CrossRef]
45. Band, N.; Kadmon, R.; Mandel, M.; DeMalach, N. Assessing the Roles of Nitrogen, Biomass, and Niche Dimensionality as Drivers of Species Loss in Grassland Communities. *Proc. Natl. Acad. Sci. USA* **2022**, *119*, e2112010119. [CrossRef]
46. Afzal, M.R.; Naz, M.; Ashraf, W.; Du, D. The Legacy of Plant Invasion: Impacts on Soil Nitrification and Management Implications. *Plants* **2023**, *12*, 2980. [CrossRef] [PubMed]
47. Li, S.; Liu, Y.; Wang, J.; Yang, L.; Zhang, S.; Xu, C.; Ding, W. Soil Acidification Aggravates the Occurrence of Bacterial Wilt in South China. *Front. Microbiol.* **2017**, *8*, 00703. [CrossRef] [PubMed]
48. Msimbira, L.A.; Smith, D.L. The Roles of Plant Growth Promoting Microbes in Enhancing Plant Tolerance to Acidity and Alkalinity Stresses. *Front. Sustain. Food Syst.* **2020**, *4*, 00106. [CrossRef]
49. Dal Molin, S.J.; Ernani, P.R.; Gerber, J.M. Soil Acidification and Nitrogen Release Following Application of Nitrogen Fertilizers. *Commun. Soil Sci. Plant Anal.* **2020**, *51*, 2551–2558. [CrossRef]
50. Isbell, F.; Reich, P.B.; Tilman, D.; Hobbie, S.E.; Polasky, S.; Binder, S. Nutrient Enrichment, Biodiversity Loss, and Consequent Declines in Ecosystem Productivity. *Proc. Natl. Acad. Sci.* **2013**, *110*, 11911–11916. [CrossRef] [PubMed]
51. Meyer, S.E.; Callaham, M.A.; Stewart, J.E.; Warren, S.D. Invasive Species Response to Natural and Anthropogenic Disturbance. In *Invasive Species in Forests and Rangelands of the United States*; Springer International Publishing: Cham, Switzerland, 2021; pp. 85–110.

52. Staude, I.R.; Pereira, H.M.; Daskalova, G.N.; Bernhardt-Römermann, M.; Diekmann, M.; Pauli, H.; Van Calster, H.; Vellend, M.; Bjorkman, A.D.; Brunet, J.; et al. Directional Turnover towards Larger-ranged Plants over Time and across Habitats. *Ecol. Lett.* **2022**, *25*, 466–482. [CrossRef] [PubMed]
53. Wang, Q.; Chen, L.; Xu, H.; Ren, K.; Xu, Z.; Tang, Y.; Xiao, J. The Effects of Warming on Root Exudation and Associated Soil N Transformation Depend on Soil Nutrient Availability. *Rhizosphere* **2021**, *17*, 100263. [CrossRef]
54. Reich, P.B.; Hobbie, S.E.; Lee, T.D. Plant Growth Enhancement by Elevated CO2 Eliminated by Joint Water and Nitrogen Limitation. *Nat. Geosci.* **2014**, *7*, 920–924. [CrossRef]
55. Brol, A.F.; Delmoral, A.P.G.; Longo, I.; Waskiewic, M.E. Variação Espaço Temporal Da Qualidade Da Água de Reservatório Em Empreendimento Hidrelétrico. In *Ciências Ambientais e da Saúde na Atualidade: Insights Para Alcançar os Objetivos Para o Desenvolvimento Sustentável*; Instituto de Inteligência em Pesquisa e Consultoria Cientifica Ltd.: Saint Paul, Brazil, 2022; pp. 79–96.
56. Oh, J.-W.; Pushparaj, S.S.C.; Muthu, M.; Gopal, J. Review of Harmful Algal Blooms (HABs) Causing Marine Fish Kills: Toxicity and Mitigation. *Plants* **2023**, *12*, 3936. [CrossRef]
57. Ogidi, O.I.; Akpan, U.M. *Aquatic Biodiversity Loss: Impacts of Pollution and Anthropogenic Activities and Strategies for Conservation*; Springer: Berlin/Heidelberg, Germany, 2022; pp. 421–448.
58. Ward, M.; Jones, R.; Brender, J.; De Kok, T.; Weyer, P.; Nolan, B.; Villanueva, C.; Van Breda, S. Drinking Water Nitrate and Human Health: An Updated Review. *Int. J. Environ. Res. Public Health* **2018**, *15*, 1557. [CrossRef]
59. Fossen Johnson, S. Methemoglobinemia: Infants at Risk. *Curr. Probl. Pediatr. Adolesc. Health Care* **2019**, *49*, 57–67. [CrossRef] [PubMed]
60. Chadwick, D.R.; Williams, J.R.; Lu, Y.; Ma, L.; Bai, Z.; Hou, Y.; Chen, X.; Misselbrook, T.H. Strategies to Reduce Nutrient Pollution from Manure Management in China. *Front. Agric. Sci. Eng.* **2020**, *7*, 45. [CrossRef]
61. Adla, K.; Dejan, K.; Neira, D.; Dragana, Š. Degradation of Ecosystems and Loss of Ecosystem Services. In *One Health*; Elsevier: Amsterdam, The Netherlands, 2022; pp. 281–327.
62. Howarth, R.W.; Marino, R. Nitrogen as the Limiting Nutrient for Eutrophication in Coastal Marine Ecosystems: Evolving Views over Three Decades. *Limnol. Oceanogr.* **2006**, *51*, 364–376. [CrossRef]
63. Rajan, M.; Shahena, S.; Chandran, V.; Mathew, L. Controlled Release of Fertilizers—Concept, Reality, and Mechanism. In *Controlled Release Fertilizers for Sustainable Agriculture*; Elsevier: Amsterdam, The Netherlands, 2021; pp. 41–56.
64. Rai, P.K. *Phytoremediation of Emerging Contaminants in Wetlands*; CRC Press: Boca Raton, FL, USA, 2018; ISBN 9781351067430.
65. Isik, S.; Haas, H.; Kalin, L.; Hantush, M.M.; Nietch, C. Nutrient Removal Potential of Headwater Wetlands in Coastal Plains of Alabama, USA. *Water* **2023**, *15*, 2687. [CrossRef] [PubMed]
66. Manale, A.P. New Policy Directions. In *Nitrogen in the Environment*; Elsevier: Amsterdam, The Netherlands, 2008; pp. 649–684.
67. Ladha, J.K.; Peoples, M.B.; Reddy, P.M.; Biswas, J.C.; Bennett, A.; Jat, M.L.; Krupnik, T.J. Biological Nitrogen Fixation and Prospects for Ecological Intensification in Cereal-Based Cropping Systems. *Field Crops Res.* **2022**, *283*, 108541. [CrossRef] [PubMed]
68. Zhang, X.; Davidson, E.A.; Mauzerall, D.L.; Searchinger, T.D.; Dumas, P.; Shen, Y. Managing Nitrogen for Sustainable Development. *Nature* **2015**, *528*, 51–59. [CrossRef] [PubMed]
69. Horák, J.; Kotuš, T.; Toková, L.; Aydın, E.; Igaz, D.; Šimanský, V. A Sustainable Approach for Improving Soil Properties and Reducing N2O Emissions Is Possible through Initial and Repeated Biochar Application. *Agronomy* **2021**, *11*, 582. [CrossRef]
70. Dawar, K.; Fahad, S.; Jahangir, M.M.R.; Munir, I.; Alam, S.S.; Khan, S.A.; Mian, I.A.; Datta, R.; Saud, S.; Banout, J.; et al. Biochar and Urease Inhibitor Mitigate NH3 and N2O Emissions and Improve Wheat Yield in a Urea Fertilized Alkaline Soil. *Sci. Rep.* **2021**, *11*, 17413. [CrossRef] [PubMed]
71. Abalos, D.; Jeffery, S.; Sanz-Cobena, A.; Guardia, G.; Vallejo, A. Meta-Analysis of the Effect of Urease and Nitrification Inhibitors on Crop Productivity and Nitrogen Use Efficiency. *Agric. Ecosyst. Environ.* **2014**, *189*, 136–144. [CrossRef]
72. Govindasamy, P.; Muthusamy, S.K.; Bagavathiannan, M.; Mowrer, J.; Jagannadham, P.T.K.; Maity, A.; Halli, H.M.; Sujayanand, G.K.; Vadivel, R.; Das, D.K.; et al. Nitrogen Use Efficiency—A Key to Enhance Crop Productivity under a Changing Climate. *Front. Plant Sci.* **2023**, *14*, 1121073. [CrossRef]
73. Matczuk, D.; Siczek, A. Effectiveness of the Use of Urease Inhibitors in Agriculture: A Review. *Int. Agrophysics* **2021**, *35*, 197–208. [CrossRef]
74. Guo, Y.; Naeem, A.; Becker-Fazekas, S.; Pitann, B.; Mühling, K.H. Efficacy of Four Nitrification Inhibitors for the Mitigation of Nitrous Oxide Emissions under Different Soil Temperature and Moisture. *J. Plant Nutr. Soil Sci.* **2022**, *185*, 60–68. [CrossRef]
75. Lu, Y.; Li, D.; Li, C.; Sun, M.; Wu, Z.; Sun, Z. The Effects of 3,5-Dimethylpyrazole on Soil Nitrification and Related Enzyme Activities in Brown Soil. *Agronomy* **2022**, *12*, 1425. [CrossRef]
76. Ren, B.; Guo, Y.; Liu, P.; Zhao, B.; Zhang, J. Effects of Urea-Ammonium Nitrate Solution on Yield, N_2O Emission, and Nitrogen Efficiency of Summer Maize Under Integration of Water and Fertilizer. *Front. Plant Sci.* **2021**, *12*, 700331. [CrossRef]
77. Dattamudi, S.; Wang, J.J.; Dodla, S.K.; Viator, H.P.; DeLaune, R.; Hiscox, A.; Darapuneni, M.; Jeong, C.; Colyer, P. Greenhouse Gas Emissions as Influenced by Nitrogen Fertilization and Harvest Residue Management in Sugarcane Production. *Agrosystems Geosci. Environ.* **2019**, *2*, 1–10. [CrossRef]
78. Ren, B.; Huang, Z.; Liu, P.; Zhao, B.; Zhang, J. Urea Ammonium Nitrate Solution Combined with Urease and Nitrification Inhibitors Jointly Mitigate NH_3 and N_2O Emissions and Improves Nitrogen Efficiency of Summer Maize under Fertigation. *Field Crops Res.* **2023**, *296*, 108909. [CrossRef]

79. Nikolajsen, M.T.; Pacholski, A.S.; Sommer, S.G. Urea Ammonium Nitrate Solution Treated with Inhibitor Technology: Effects on Ammonia Emission Reduction, Wheat Yield, and Inorganic N in Soil. *Agronomy* **2020**, *10*, 161. [CrossRef]
80. Peng, H.; Xu, T.; Wang, L.; Yu, J.; Chen, X.; Cheng, X.; Li, H.; Huang, L.; Wei, L.; Wei, S. Effect of Streptomyces JD211 Application on Soil Physicochemical Properties and N_2O Emission Characteristics of Rice Rhizosphere. *Sci. Total Environ.* **2024**, *906*, 167673. [CrossRef]