Chapter 13
Nitrogen Regulation in China's Agricultural Systems

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Abstract Nitrogen (N) fertilizer has played an important role in maintaining grain production. The amount of crop N uptake doubled from 1980 to 2010, with total N input increasing more than three times in the Chinese agricultural system during the same period. High input has resulted in high output and surplus. More than 50% N surplus can cause environmental problems and consequently threaten human health and ecosystems services. Improving N use efficiency (NUE) is critical to maintain crop productivity and environmental sustainability. However, historical NUE in croplands in China decreased from 1980 to 2010. Several research efforts have been conducted to improve NUE in China, such as the integrated soil-crop system management (ISSM), knowledge-based N management, and livestock manure partially substituting for synthetic fertilizer. Moreover, China's government has created a series of programs, projects, and regulations to enhance NUE and reduce environmental risk, such as the soil testing and fertilizer recommendation (STFR) program and "Zero Increase Action Plan for Fertilizer Use." These technologies, policies, and management practices can significantly increase crop yield and decrease reactive N (Nr) losses. However, more research focused on integrated technology and management systems combined with socioeconomic is still needed in China in the future.

13.1 Introduction

As a key fundamental nutrient, nitrogen (N) plays an important role in sustaining life. It can sustain food production and global population after it is converted into reactive N (Nr) species (Galloway et al. 2004; Mueller et al. 2012). To meet global food demands, agriculture consumes greater than 100 Tg N year^{-1} of Nr, accounting for more than 50% of the total annual terrestrial Nr (Sutton et al. 2013). A large amount of N input to cropland has increased crop production by more than 40% throughout the world (Li et al. 2009, Malhi et al. 2001). Nearly 50% of the world

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population has been fed by the application of chemical N fertilizer (Erisman et al. 2008). Given the increased food and biofuel demands of the growing world population, global crop production in 2050 must double compared to that of 2005; thus, the consumption of N fertilizer could increase by as much as 2.7 times (Tilman et al. 2011; Tilman et al. 2002).

Indeed, N plays an important role in increasing grain production that provides humans' food including calories and proteins. However, only 10% of the N used for food production is consumed by the human beings (Robertson and Vitousek 2009). On average, the global crop N use efficiency (NUE) is only approximately 42–47% (Mueller et al. 2017; Zhang et al. 2015). A substantial fraction of Nr that is lost into the environment through emission, runoff, etc. can cause enormous adverse environmental problems such as water pollution, climate forcing, biodiversity loss, and air pollution (Galloway et al. 2004; Zhang et al. 2015; Yu et al., 2019). Furthermore, N pollution costs between €70 billion and €320 billion per year in the European Union based on the damage of Nr on human health and ecosystems (Sutton et al. 2011a). Therefore, increasing crop production while decreasing environmental costs and protecting human health is urgently needed for N management in agricultural systems (Lassaletta et al. 2016; Li et al. 2017b; Sutton et al. 2011b; Tilman et al. 2002).

As the world's largest developing country, China has applied approximately 30% of the global annual production of chemical N fertilizer for crop production in croplands (Hou et al. 2013; Huang et al. 2017). China's grain production has doubled over the last several decades, from 305Mt in 1978 to 621Mt in 2015 (National Bureau of Statistics of China 2016). The total chemical N fertilizer use increased from 9.4 Tg to 29.5 Tg from 1980 to 2010 (Yan et al. 2014). Per hectare N addition was 209, 210, and 220 kg N ha^{-1} for rice, wheat, and corn, respectively, in recent years (Chen et al. 2014). And there were much higher N application rates for intensive double-cropping systems (550–600 kg N ha^{-1}) and some greenhouse vegetables, >1000 kg N ha^{-1}, compared to the major crops (Ju et al. 2009; Shi et al. 2009).

A large amount of N fertilizer applied to croplands plays a vital role in ensuring China's food security. The relationship between the application of fertilizer and food production is significant (Jiao et al. 2016; Zhang et al. 2012a; Zhu and Chen 2002). However, excessive N use has resulted in a low NUE. The in-season NUE was 30–35% in the 1990s in China from field monitoring results (Zhu and Chen 2002), and it decreased to 26–28% in 2001–2005 (Zhang et al. 2007), which was much lower than that in America and Europe (Ladha et al. 2005). Much of the N in the form of ammonia, nitrate, and N oxides is lost to the environment, which are considered expensive and environmentally damaging waste in China (Guo et al. 2010; Liu et al. 2013; Wang et al. 2016; Wang and Zhao 2014).

Presently, environmental sustainability is a policy priority in China; however, China needs to produce more grain yield to meet the demand of an increasing population in the near future, along with a change in diet and energy needs (Miao et al. 2011). Therefore, regulations for N to optimize benefits for food production while minimizing damage to human health and ecosystem pollution from

agricultural systems in China must be addressed. Studies regarding the application of N fertilizer and its contributions to environmental pollution could support basic information for N regulation (Reis et al. 2016). Therefore, from this view, we (1) gave an overview of N use and fates in China; (2) analyzed methods to improve NUE in China's agricultural systems; and (3) summarized the effects of practices on mitigating N loss and provided possible recommendations for future agricultural N regulation in China.

13.2   Nitrogen Use in China's Agricultural Systems

13.2.1   Nitrogen Input to China's Agricultural System

Major N input to China's agricultural system includes synthetic N such as urea and compound fertilizer and non-synthetic N such as biological fixation, atmospheric deposition, and organic fertilizer. At a national scale, Yan et al. (2014) estimated that total N inputs in Chinese croplands increased from 17.3 Tg year^{-1} (142 kg ha^{-1}) in 1980 to 42.0 Tg year^{-1} (322 kg ha^{-1}) in 2010 (Table 13.1), with a sharp increase of 142.8%.

Organic materials such as manure have always been reused as fertilizer in China. The application of organic manures maintains soil fertility and crop yield (Cui et al. 2014a; Ju et al. 2005). From 1980 to 2010, the amount of organic N fertilizer (including human and livestock manure and crop residues) input to croplands of China increased from 3.7 Tg year^{-1} to 5.7 Tg year^{-1}, while the percentage of organic N fertilizer of the total N input decreased from 21.4% to 13.6% (Fig. 13.1). Factors such as income growth and improvement in rural waste treatment have resulted in a significant declining use of human manure (Liu et al. 2014b). Because of the significant importance of natural processes in world agriculture, biological N fixation is another source of N input in agricultural systems (Herridge et al. 2008). N$_2$ fixation in Chinese croplands slightly changed during the 1980–2010

Table 13.1 Total crop N uptake and N input to croplands of China[*]

| Year | Crop N uptake (Tg N year^{-1}) | N input to croplands (Tg N year^{-1}) | | | |
		Synthetic fertilizer	Organic fertilizer	Biological fixation	Atmospheric deposition
1980	8.1 (62)	9.4 (72)	3.7 (28)	2.8 (21)	1.4 (11)
1985	10.5 (80)	12.6 (96)	4.4 (34)	2.6 (20)	1.9 (14)
1990	12.0 (92)	17.4 (133)	5.0 (38)	2.6 (20)	2.4 (18)
1995	13.4 (103)	22.3 (171)	6.0 (46)	3.0 (23)	2.6 (20)
2000	14.1 (108)	24.6 (189)	6.1 (47)	3.1 (24)	2.8 (22)
2005	15.2 (117)	26.6 (205)	6.6 (51)	3.0 (23)	3.1 (24)
2010	16.9 (139)	29.5 (242)	5.7 (47)	2.9 (24)	3.9 (32)

The unit of data in brackets is kg N per hectare of arable land area
[*]This table was adapted from Yan et al. (2014)
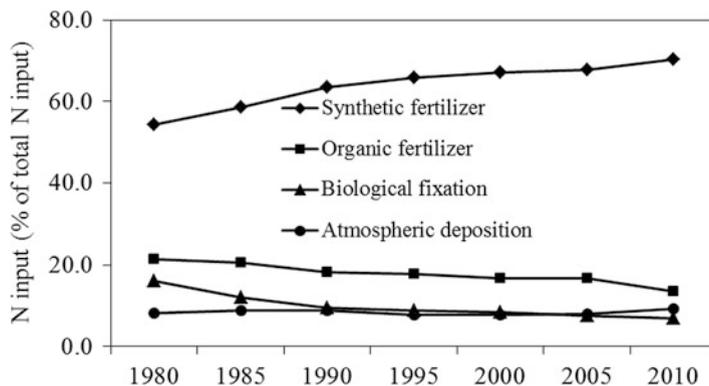
Fig. 13.1 Percentages of N input sources from 1980 to 2010 in China's croplands

period. Similarly, as in the use of organic fertilizer, the percentage of biological N fixation sharply decreased from 1980 to 2010.

In contrast, the contributions from both chemical fertilizers and atmospheric deposition have increased during the past three decades. Annual domestic production and consumption of chemical fertilizer has largely increased because of the expansion of the fertilizer industry and the support of a series of policies and subsidy program established during the 1990s in China (Li et al. 2013b). The use of chemical N fertilizer increased by 213.8% from 1980 to 2010. Chemical fertilizer accounted for 70.2% of the total N input in 2010 (Fig. 13.1). Furthermore, atmospheric N deposition increased by 178.6% during this period as a result of the rapid agricultural, industrial, and urban development (Liu et al. 2013).

13.2.2 Spatial and Temporal Variations of N Inputs

The use of N in China's agricultural systems has varied among provinces. Total N inputs were higher in the southeast, central, and North China Plain, such as in Jiangsu, Zhejiang, Hubei, and Shandong provinces, compared to western and northern provinces such as Tibet, Gansu, Xinjiang, and Inner Mongolia (Sun et al. 2008). Additionally, from Li et al. (2013a), chemical N fertilizer application and the total amount of N input were higher in the north central and middle/lower Yangtze River regions. Chemical N fertilizer used in these regions accounted for 56.4% of the total N fertilizer input in China. Gu et al. (2017) showed that N input in northwest China was lower than that in southeast China.

Obviously, total N input in provinces/regions with relatively high per capita gross domestic product (GDP) and intensive human population was high. This result indicated that the demand for food, fiber, and energy was affected by the economic growth. The N budget of mainland China showed that the total N input was highly correlated with population density and per capita GDP (Ti et al. 2012). For example,

national statistical data (National Bureau of Statistics of China 2016) showed that per capita GDPs in Beijing, Shanghai, Jiangsu, and Zhejiang were 106,497, 103,896, 87,995, and 77,644 RMB yuan (¥) in 2015, respectively, which were much higher than those reported in Gansu (26165 ¥) and Xinjiang (40648 ¥).

13.2.3   Nitrogen Application on Cereals and Vegetables

Globally, approximately 50% of Nr produced by the Haber-Bosch process is consumed by major cereals including wheat (*Triticum aestivum* L.), maize (*Zea mays* L.), and rice (*Oryza sativa* L.) (Ladha et al. 2016). From farmer surveys and data mining, N fertilizer application on maize, wheat, and rice consumed approximately 15%, 18%, and 21%, respectively, of the total N fertilizer input to Chinese farmlands during 2000–2003 (Wu et al. 2016). The amount of N fertilizer applied to maize increased by 4% during 2008–2009, with a decrease in application on rice and wheat (Wu et al. 2016). On average, the N application on wheat, maize, and rice increased sharply from 45–60 kg N ha^{-1} during 1970–1979 to 200–300 kg N ha^{-1} in 2010 (Cui et al. 2014a; Wu et al. 2016). Additionally, the N fertilizer application rate of wheat was higher in the North China Plain, while the highest N input rate of rice occurred in Yangtze River region.

Furthermore, the cultivated land area of vegetables has increased nearly six times from 1978 to 2011, resulting in a significant increase in vegetable production. The vegetable planted area accounted for 12.1% of the total national crop-sown area in 2011 (National Bureau of Statistics of China 2016). Vegetable fields are often characterized by intensive crop rotations (multiple harvests within a single year); thus, more N fertilizer is applied to vegetables compared to maize, wheat, and rice. For example, the average annual N input including both chemical fertilizers and manure was 3239 kg N ha^{-1} in some vegetable greenhouses in northern China (Ju et al. 2006). Our results (Ti et al. 2015) showed that total N input in vegetable cropping system in 2010 was 8.04 Tg N year^{-1}. Chemical fertilizer N input in the greenhouse system (478 kg N ha^{-1} per season) was much higher than that in the open-air system (201 kg N ha^{-1} per season).

Fruits and tea also consumed a large amount of N fertilizer. Nitrogen applied to fruit and tea farmland accounted for 13% and 17% of the total N input on Chinese farmland during the 2000–2003 and 2008–2009 periods, respectively. The rate of chemical N fertilizer use on conventional fruit and tea farmland was 330 and 536 kg N ha^{-1}, respectively, which was much higher than that of maize, rice, and wheat (Meng et al. 2017). Overall, compared to other regions/countries, such as the USA and the EU, N use in China's agricultural systems is much higher (Zhang et al. 2015).

13.3   Nitrogen Loss in China's Agricultural Systems

The characteristics of Chinese agricultural N balance are high input, output, and surplus (Chen et al. 2016). In addition, the application rate of chemical fertilizer is high. The input of N and cereal production have continued to increase, while the growth rate of recent crop yields has slowed (Li et al. 2016). Because of the high output and surplus, a large amount of N is lost into the environment via NH_3 and N_2O/NO emission, leaching, runoff, etc. A summary study on soil N in China indicated that N losses from cropland through NH_3 volatilization, nitrification, and denitrification, and leaching and runoff account for 11%, 34%, 2%, and 5% of total N use, respectively.

Ammonia volatilization is an important means of gaseous NH_3 loss from the croplands. The NH_3 emission from soil is a complex process affected by physical, chemical, and biological factors (Bussink and Oenema 1998; Haynes and Williams 1993). Basically, the pH of the soil solution plays an important role in NH_3 loss, which can be explained by the following equation:

$$NH_4^+ + OH^- \leftrightarrow NH_3 + H_2O \qquad (13.1)$$
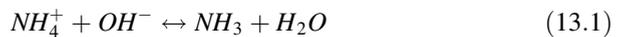
Urea hydrolysis produces highly concentrated NH_4^+ with a sharply increased pH and then in turn dissociates to produce NH_4^+, NH_3, and OH^-, as described by Cameron et al. (2013) in the following:

$$(NH_2)_2CO + 2H_2O \rightarrow (NH_4)_2CO_3 \rightarrow NH_4^+ + NH_3 \uparrow + CO_2 + OH^- \qquad (13.2)$$
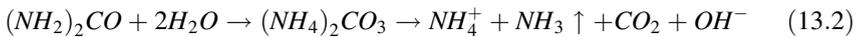
Factors including soil pH, temperature, ammonium concentration, soil cation exchange capacity and moisture, rainfall, irrigation, fertilizer use, and even plant and field management affect the NH_3 volatilization rate and amount (Black et al. 1985; Mcgarry et al. 1987; Sommer et al. 2004; Turner et al. 2012; Whitehead and Raistrick 1993).

N_2 and N_2O are the dominant forms of gaseous N and are also significantly lost from soil/plant systems following N fertilizer application. From the processes of nitrification and denitrification, N_2 and N_2O are emitted to the air. Under soils with anaerobic, low oxygen availability and low redox conditions, biological denitrification can occur (Cameron et al. 2013). The process of denitrification can be illustrated by the following equation:

$$2NO_3^- \rightarrow 2NO_2^- \rightarrow 2NO \rightarrow N_2O \rightarrow N_2 \qquad (13.3)$$
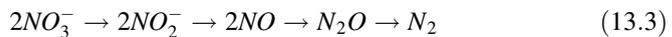
Reductases such as NO_3^- reductase, NO_2^- reductase, and N_2O reductase are enzymes responsible for reduction (Cameron et al. 2013). Furthermore, chemical reactions can produce N_2 and/or N_2O, and fungal denitrification can also occur in soils (Gupta et al. 2011; Hayatsu et al. 2008; Thomson et al. 2012). In addition, denitrification can be affected by soil moisture, aeration, nitrate and ammonium,

carbon, soil pH, and temperature (Bouwman 1996; Bouwman et al. 1993, Freney 1997; Hayatsu et al. 2008; Thomson et al. 2012).

Among the main flow pathways that contribute to N losses from agricultural systems is N runoff and leaching. The characteristics of soil type, organic matter, and particle-size distribution can affect the amount of N runoff from soils (Dreelin et al. 2006; Ouyang et al. 2010; Zhang et al. 2012b). Factors such as the terrain condition, rainfall intensity, vegetation fraction, and soil moisture content can also affect the N runoff (Bakhsh et al. 2005; B ouldin et al. 2004; Udawatta et al. 2006). Moreover, the N loss through runoff can be affected by fertilization (Liu et al. 2014a). Similarly, the amount of N leaching from agricultural systems also depends on the soil characteristics, crop type, climatic conditions that dominated leaching, fertilizer application rate, and field management practices (Goulding 2000; Schepers et al. 1995; Westermann et al. 1988).

With a large amount of N input, aboveground crop N uptake from croplands of China increased from 8.1 to 16.9 Tg N year^{-1} from 1980 to 2010 (Yan et al. 2014), which accounts for less than 50% of the total N input. Therefore, decades of fertilizer N overuse have caused a large amount of N to be lost to the soil, water, and air. N loss from volatilization and denitrification increased from 3.1 ± 1.0 to 7.6 ± 2.5 and from 3.1 ± 1.3 to 7.3 ± 3.0 Tg N year^{-1} from 1980 to 2010. Approximately 2.1 and 2.4 Tg N year^{-1} was lost through leaching and surface runoff, as well as 7.5 ± 4.9 Tg N year^{-1} accumulated in the soil. N surplus in croplands in China increased 1.4 times (86.3–226.9 kg N ha^{-1} year^{-1}) in the same period (Gu et al. 2017).

For major crops, Cui et al. (2014b) indicated that the average loss of N_2O from rice, wheat, and maize in China was 0.70%, 0.53%, and 0.86% of N fertilizer application, respectively. The loss from leaching was accounted for 2.98%, 10.1%, and 22.2% of the N application. NH_3 volatilization from the aforementioned crops was as high as 17%, 14%, and 24% of the total N input from rice, wheat, and maize, respectively. Furthermore, given the intensive crop rotations and high N application rates, approximately 0.32, 0.02, and 0.86 Tg N year^{-1} were lost through leaching, NH_3 emission, and nitrification and denitrification, respectively, from the Chinese greenhouse vegetable cropping system. However, the amounts of N losses from the open-air vegetable cropping system were much higher than those from the greenhouse vegetable cropping system (Ti et al. 2015).

The overuse of N fertilizer has contributed to environmental pollution and costly damage in China. For example, N losses through NH_3 and N_2O emissions are responsible for the heavy N deposition and air pollution in China (Liu et al. 2013; Reis et al. 2009). Ti and Yan (2013) pointed out that 22–57% of the total nonpoint N originated from cropland fertilizer runoff transported in large river basins in China causing water pollution. Moreover, a life-cycle analysis of the three major crop productions in China showed that the total damage costs of GHG emission and Nr N loss were equal to 1.44% of the Chinese GDP (Xia et al. 2016).

13.4   Methods to Improve NUE in China's Agricultural Systems

13.4.1   *Nitrogen Use Efficiency of China's Agricultural Systems*

Generally, NUE can be express as the yield produced per unit of N applied (Cui et al. 2014b). Although large amount of N fertilizer has been applied to croplands, less than 50% of the Nr input is converted into harvested products, compared to that during the early 1960s (Lassaletta et al. 2014). Moreover, field experiments have shown that the percentage of in-season N fertilizer recovery ranged from 30% to 35% in the 1990s in China (Zhu and Chen 2002). NUE has varied throughout history in China. It increased from $40 \pm 3\%$ to $45 \pm 3\%$ from 1980 to 1984, decreased to $36 \pm 2\%$ in 2003, and increased to $39 \pm 2\%$ in 2010 (Gu et al. 2017). Specifically, the NUE of vegetable cropping systems was lower than the average NUE of the three major crops. For example, the NUE of an open-air system was $25.9 \pm 13.3\%$ (Ti et al. 2015). The kilogram of harvested product per kilogram of N applied for the three major crops markedly decreased from 1979 to 2008 (Wu et al. 2016).

13.4.2   *Measures to Increase NUE*

Improving NUE is one of the most effective ways to increase crop productivity while reducing environmental degradation (Zhang et al. 2015). China's NUE was lower than that in other regions such as North America and Europe (Ladha et al. 2005; Lassaletta et al. 2014; Zhang et al. 2015). Achieving food and environmental security in the near future is imperative and urgent in China. Therefore, it is important to improve NUE in China. Furthermore, Zhang et al. (2015) indicated that China's NUE still has not passed the turning point of the environmental Kuznets curve, which means that it is possible to increase NUE in China by using N reduction mitigations.

Farm-scale technologies and management practices are required to improve the NUE in China. Strategies such as improvements in irrigation and fertilizer management and the use of inhibitors could reduce N loss and improve crop productions. Furthermore, regular attention should be paid to establishing national and local targets and policies. Measures to improve NUE have been widely conducted in China. Measures such as the integrated soil-crop system management (ISSM) led by China Agricultural University could significantly increase the crop grain yield by 18.1–35.2%, while reducing N application rate by 3.8–14.4% compared to farmer's current practice, which indicates a higher improvement in NUE (Chen et al. 2014). Additionally, knowledge-based N management, including controlled-release N fertilizer application, nitrification and urease inhibitors, higher N fertilizer application splitting frequencies, lower basal N fertilizer proportions, deep placement of N

fertilizer, and an optimal N rate based on soil N tests, could significantly increase crop production and reduce Nr losses in China (Xia et al. 2017a). Even the simple practice of using a urease inhibitor (e.g., Limus) application can lead to a yield increase of 0–5%, NUE increase of 13–14%, and NH_3 loss decrease by 80–84% (Li et al. 2015, 2017a). Otherwise, application of livestock manure in croplands to substitute chemical N fertilizer significantly increased crop yield and annual SOC sequestration while reducing Nr losses via NH_3 emission, N leaching, and N runoff (Xia et al. 2017b).

Much research has focused on technological innovation to improve NUE in crop production, but the socioeconomic constraints are at present poorly understood. Fertilizer use on a per area basis sharply decreased with the increase in farm size, and the crop yield is higher at large-scale farms compared to that in smallholder farms in China. The high labor cost suggests a low machinery level at smallholder farms, which inhibits the application of precise fertilization technologies and management based on scientific knowledge (Ju et al. 2016). Considering the benefits of N fertilizer use to ecological and social systems based on 156 farm sites with five different N levels, agronomically, privately, ecologically, and socially optimized, Ying et al. (2017) indicated that N management using the ecologically and socially optimized N rate provides a win-win opportunity with a significant increase in grain yield while reducing Nr loss of wheat; it should be prioritized in agricultural research and practice.

In fact, to seek food and environmental security, the central government of China has officially established a series of programs, projects, and regulations to enhance NUE and reduce environmental risk in search of sustainable agriculture development. For example, a soil testing and fertilizer recommendation (STFR) program used to decrease N fertilizer application was implemented by the Ministry of Agriculture (MOA) since the late 1990s. Following implementation of the program, approximately 66.7 million hectares of croplands had been incorporated by 2009. An announcement of "Zero Increase Action Plan for Fertilizer Use" was launched in 2015; it is reasonable to be confident that the NUE for crop production in China could be further increased. According to this announcement, China plans to develop precision agriculture, adjust the fertilizer consumption structure, and use more organic fertilizer to replace chemical fertilizers. Based on these policies, research has shown that under a no more chemical fertilizer input scenario from 2020, NUE would be a few percentage points higher than it is in the current scenario, but its effect on nutrient pollution reduction was not effective in reducing nutrient pollution in China (Wang et al. 2017).

13.5   Summary/Policy Implications

The issue of N fertilizer overuse has long been recognized. N management practices and mitigate strategies should be considered when policy-makers wish to increase the NUE or decrease N fertilizer use. More regulations need to be developed and

extended to reduce the use of N fertilizer in the future, and the mitigation of the adverse impacts of agricultural N overuse on the environment and human health in China still needs attentions. Chinese farmers prefer to increase the input cost while not always considering the effectiveness of N mitigation strategies. Generally, the farm size in China is tiny; thus, it is difficult to implement mechanizations such as those used at large scales (Xia et al. 2017a). Increasing farm sizes combined with policy regulations and market exchange should be integrated to reduce N pollution while increasing crop production (Ju et al. 2016). Therefore, comprehensive research and development of innovative agricultural technologies and management systems with socioeconomic research are still needed in China. Moreover, to gain insight into the agronomic effect and environmental impact of N, a comprehensive N assessment of the agricultural system should be undertaken in the future. For example, the key driving factors of NUE and solutions for how to reduce the uncertainties and understand the mechanisms of N loss (particularly of N loss through denitrification) should be investigated.

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