Agricultural Systems 167 (2018) 125–135

Contents lists available at ScienceDirect

Agricultural Systems

journal homepage: www.elsevier.com/locate/agsy

Estimating soil nitrogen balance at regional scale in China's croplands from 1984 to 2014

Wentian He[a,1], Rong Jiang[a,1], Ping He[a,b,*], Jingyi Yang[c], Wei Zhou[a,*], Jinchuan Ma[a], Yingxia Liu[a]

[a] Key Laboratory of Plant Nutrition and Fertilizers, Ministry of Agriculture, Institute of Agricultural Resources and Regional Planning, Chinese Academy of Agricultural Sciences (CAAS), Beijing 100081, China

[b] CAAS-IPNI Joint Lab for Plant Nutrition Innovation Research, International Plant Nutrition Institute (IPNI) China Program, Beijing 100081, China

[c] Agriculture and Agri-Food Canada, Harrow Research and Development Centre, 2585 County Road 20, Harrow, ON, N0R 1G0, Canada

ARTICLE INFO

Keywords:
Nitrogen balance
Chemical fertilizer
Manure
Nitrogen losses
Nitrogen use efficiency

ABSTRACT

Estimating regional soil nitrogen (N) balance in croplands is critical to improve management practices, reduce environmental risks and develop sustainable agriculture. In this study, spatial and temporal variations of soil N balance were evaluated from 1984 to 2014 in China's croplands. Results indicated that the total soil N balance was in surplus and increased by 7.3 Tg N (130.4%) between 1984 and 2014, which was attributed to the increased N input of 29.3 Tg N, compared with the increased N output of 22.1 Tg N. Soil N balance continually increased from the 1980s (1984–1989) to the 2000s (2000–2009), and then decreased in the 2010s (2010–2014). Meanwhile, N use efficiency decreased gradually from the 1980s to the 2000s, but it increased in the 2010s. The N loss (N_2, N_2O, NO, NH_3, NO_3^- leaching and runoff) increased significantly from the 1980s to the 1990s, while the increasing trend gradually reduced from the 1990s to the 2010s. The spatial-temporal distribution of the N balance at the regional scale showed that the total highest and lowest N balance was in the middle and lower reaches of Yangtze River (2.1–3.7 Tg N) and northeast of China (0.3–1.0 Tg N), but the highest and lowest N balance per cropping area was in the southeast (93.4–129.7 kg N ha^{-1}) and northeast (19.6–43.9 kg N ha^{-1}) regions respectively from the 1980s to the 2010s. The N balance decreased for all regions from the 2000s to the 2010s, excluding the southeast and southwest of China due to higher increased rate of N input than the lower increased rate of N output. Reducing the use of chemical fertilizer N would improve cop productivity, decrease soil surplus N and environmental risks of N gas emissions, nitrate leaching and runoff.

1. Introduction

The aim of modern sustainable agriculture is to guarantee crop productivity in limited arable land to satisfy the demand of increasing population and to reduce environmental risks. The global population is expected to soar toward 9 billion by 2050, with global demand of crop production predicted to increase by about 70% (FAO, 2009; Godfray et al., 2010). In China, the amount of N application in croplands was almost fully dependent on manure fertilizer before the 1970s (excluding biological N fixation), but manure fertilizer is not sufficient for the crops nutrient demands (Wang et al. 2016). Chemical fertilizer application increased dramatically since the 1980s, and the consumption of total chemical fertilizer and N fertilizer in China accounted for 31% and 29% of the world's total in 2014, respectively (FAOSTAT 2017). Chemical fertilizer N application contributed to 30–50% of the world's crop yield (Erisman et al. 2008). Approximately 110 Tg N was applied to meet food demand and accounted for about 56% of the total fertilizer input in 2014, worldwide (FAOSTAT 2017). Increased N fertilizer input with low N use efficiency is unlikely to be as effective for yield improvement (Tilman et al. 2002). On the contrary, excessive chemical fertilizer N application induced a series of environmental problems including greenhouse gas emissions, non-point source pollution and soil acidification (Drecht et al. 2003; Guo et al. 2010; Erisman et al. 2011).

The loss of reactive N (i.e., nitrous oxide (N_2O), ammonia (NH_3) and nitrate (NO_3^-)) from agricultural systems into the environment is a major threat to the global environment. Agricultural soils have been recognized to be the main source of N_2O emission, which is one of the major greenhouse gases and is predicted to continually increase in the future (Burney et al. 2010; IPCC 2013). Direct emissions were the main contributor of the total N_2O emissions, which may result from chemical

* Corresponding authors at: Key Laboratory of Plant Nutrition and Fertilizers, Institute of Agricultural Resources and Regional Planning, Ministry of Agriculture, Chinese Academy of Agricultural Sciences (CAAS), Beijing 100081, China.

E-mail addresses: heping02@caas.cn (P. He), zhouwei02@caas.cn (W. Zhou).

[1] Wentian He and Rong Jiang contributed equally to this work.

https://doi.org/10.1016/j.agsy.2018.09.002
Received 24 August 2017; Received in revised form 7 September 2018; Accepted 9 September 2018

fertilizers, manure waste, biological N fixation and crop residues (IPCC 1996). Chemical fertilizer N is a major promoter of nitrification and denitrification to emit N_2O, by providing substrate for soil microbial activities (Snyser et al. 2009; Jones et al. 2014). Chemical fertilizer as well as manure from humans and livestock are also primary sources that generate the loss of ammonia volatilization into the atmosphere (Bouwman et al. 2002; Soares et al. 2012; Huang et al. 2016), which then feeds back into the soil through N deposition (Behera et al. 2013). Ammonium-containing aerosols are a source fine particulate matter (PM2.5), causing air pollution and harm human health (Huang et al. 2011). Additionally, due to the low N use efficiency by crops and substantial amounts of N fertilizer left in the soil, N could be lost from agricultural systems into groundwater, river and lakes though NO_3^- leaching and runoff, resulting in water eutrophication and groundwater contamination (Tilman et al. 2002; Seitzinger et al. 2010; Wang et al. 2014). Therefore, estimating soil N balance in croplands is critical for improving agricultural management practices to ensure food security and maintaining environmental health.

Nitrogen balance is mainly computed as the difference between N input and output in an agricultural system (Oenema et al. 2003; Yang et al. 2007). The Organization for Economic Co-operation and Development (OECD) also calculated gross soil N balance for the member countries using the same suites of N balance methods as an important index, to measure the condition of the agricultural ecological environment (OECD 2002). Farm-gate, soil surface and soil system are the dominating methods of budgeting nutrients in agriculture systems worldwide (Oenema et al. 2003; Hoang and Alauddin 2009). Hoang and Alauddin (2009) showed that the mean value of soil N balances for all countries was 67 kg N ha^{-1} and the average balance was 23 kg N ha^{-1} from 1985 to 2003 in OECD countries based on farm level. Additionally, the N balance in Belgium and Luxembourg, Japan and the Netherlands surpassed 160 kg N ha^{-1}. Panten et al. (2009) showed that the average soil surface N balance was about 100 kg N ha^{-1} from 1992 to 2006 in Germany. Soil system pathways can fully reflect the loss of N into the environment via volatilization, denitrification, leaching and runoff, compared to soil surface method (Oenema et al. 2003; Roy et al. 2003). The evaluation model was updated according to the actual N flow of agricultural nutrients at the national scale. A lot of researchers from different countries have evaluated soil N cycling in croplands based on nutrient budget methods at the national level, worldwide, over past decades (Stoorvogel et al. 1993; Sheldrick et al. 2002; Shindo et al. 2003; OECD 2013). The CANB Model (Canadian Agricultural Nitrogen Budget) was used to estimate residual soil N, NO_3^- leaching and N gaseous emissions (N_2O and NH_3) at a large scale (1:1 M soil mapping system) in Canada, and results from this study indicated that on average, soil residual N was 25 kg N ha^{-1} in 2001, which decreased to 17 kg N ha^{-1} in 2006 (Yang et al. 2007, 2013).

In China, various models and methods were used to estimate N balance in croplands based on the core principle of nutrient balance, such as URCNC, Eubolism and DSS models at various regional and national scales (Liu et al. 2009; Gu et al. 2015; Chen et al. 2016). Despite the differences in evaluation methods and parameters resulting in certain discrepancies, the results of the national soil N balance showed a surplus after 1980 (Liu et al. 2009; Li and Jin 2011; Cui et al. 2013; Gu et al. 2015; Chen et al. 2016) in China's croplands, excepting a fluctuated result of a small annual surplus and deficit from 1980 to 1997 (Sheldrick et al. 2003).

However, previous studies mainly focused on estimating the variation of the N flow in croplands for one year at regional level (Li and Jin 2011; Li et al. 2011; Wang et al. 2014), or in multi-year trend at the national scale before 2010 (Sheldrick et al. 2003; Liu et al. 2009; Cui et al. 2013; Gu et al. 2015). Therefore, it is urgent to spatially and temporally update and assess soil N balance, which can be used to provide scientific guidance for regional N management in an agroecosystem. The objective of this study was to establish an N balance model using soil system method, which then can evaluate regional and temporal variations of soil N balance and N use efficiency in China's croplands from 1984 to 2014.

2. Materials and methods

2.1. Data sources

The datasets used for evaluating N budget of cropland in national and regional scales mainly originate from the China Agriculture Statistical Report from 1984 to 2014 (Ministry of Agriculture PRC, 1984–2014), the National Bureau of Statistics of the People's Republic of China (http://www.stats.gov.cn/) and the China Agricultural Products Cost-benefit Compilation of Information. Based on data availability, the time periods of this study were divided into the following timelines: the 1980s (1984–1989), 1990s (1990–1999), 2000s (2000–2009) and 2010s (2010–2014). A dataset of more than ninety parameters were collected as initial inputs, including the cropping area and production, chemical fertilizer N (single and compound) as well as the population and livestock storages. The amount of cropping area was shown in Fig. S1. The missing values were interpolated using the data from adjacent years, referring to other statistical yearbooks and published reports or references.

In this study, six agricultural regions were grouped based on geographical locations and China's administrative divisions, which enabled us to evaluate regional variation of soil N balance and environment risks (i.e., N gases emissions from N_2O, NH_3, NO_3^- leaching and runoff). The six regions considered in this analysis are northeast (NE), north central (NC), the middle and lower reaches of Yangtze River (MLYR), northwest (NW), southeast (SE) and southwest (SW) of China (Table S1). The cropland for main grain and cash crops is composed of 15 soil types (grasslands and pastures were not included). In this study (Table S1), detailed descriptions were provided by Li and Jin (2011) and He et al. (2015).

2.2. Soil nitrogen balance model

The soil nitrogen balance model was based on a general soil N budget, which is the difference between N input and N output. The N input is calculated similar to OECD (2002) and Yang et al. (2007, 2013), with some modifications of input and output components at the regional scale of China's croplands (Fig. 1). The main equations are shown in Eq. (1)–(3) as below:

$$N_{balance} = N_{input} - N_{output} \tag{1}$$

$$N_{input} = N_{fert} + N_{man} + N_{cake} + N_{straw} + N_{fix} + N_{depo} + N_{irri} + N_{seed} \tag{2}$$

$$N_{output} = N_{crop\_removal} + N_{N2} + N_{N2O} + N_{NO} + N_{NH3} + N_{NO3-leaching} + N_{NO3-runoff} \tag{3}$$

$N_{balance}$ is equal to N input minus output. N_{input} represents the amount of N input to the soil, which includes total N from chemical fertilizer (N_{fert}), manure fertilizer from humans and livestock (N_{man}), cake fertilizer returns to soil (N_{cake}), straw nitrogen returns to soil (N_{straw}), N fixation from symbiotic and non-symbiotic (N_{fix}), N dry and wet deposition from the atmosphere (N_{depo}), N irrigation from irrigation water (N_{irri}) and N from the seed (N_{seed}). N_{output} includes N removal from crops ($N_{crop\_removal}$), N_2 (N_{N2}), N_2O (N_{N2O}), NO (N_{NO}) emissions and ammonia volatilization (N_{NH3}) and NO_3^- leaching ($N_{NO3\text{-}leaching}$) and runoff ($N_{NO3\text{-}runoff}$). The detailed equations for each component are summarized in Tables 1 and 2. The result of $N_{balance}$ can be interpreted as follows:

- If $N_{balance} = 0$, there is no N surplus in the soil at an annual budget.
- If $N_{balance} < 0$, it indicates a potential N stress in the soil. Thus, a N addition is needed to meet crop N requirement.
- If $N_{balance} > 0$, it indicates a potential N surplus in the soil. Thus, it may mean low N use efficiency or economic losses of N for farmers.