

Nitrogen balances of smallholder farms in major cropping systems in a peri-urban area of Beijing, China

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Abstract An in-depth understanding of nutrient management variability on the regional scale is urgently required due to rapid changes in cropping patterns and farmers' resource use in peri-urban areas of China. The soil surface nitrogen (N) balances of cereal, orchard and vegetable systems were studied over a 2-year period on smallholder fields in a representative peri-urban area of Beijing. Positive soil surface N balances were obtained across all three cropping systems. The mean annual N surplus of the vegetable system was $1,575 \text{ kg N ha}^{-1} \text{ year}^{-1}$, or approximately 3 times the corresponding values in the cereal ($531 \text{ kg N ha}^{-1} \text{ year}^{-1}$) and orchard systems ($519 \text{ kg N ha}^{-1} \text{ year}^{-1}$). In the vegetable system, animal manure ($1,443 \text{ kg N ha}^{-1} \text{ year}^{-1}$ on average) was the major source of N input (65 % of the total N input) and the factor with strongest impact on the N surplus. In the cereal system, however, about 74 % of the total N input originated from mineral fertilizer application which was the major contributor to the N surplus, while in the orchard system, the N surplus was strongly and positively correlated with both mineral

fertilizer and animal manure applications. Furthermore, within each cropping system, N fertilization, crop yields and N balances showed large variations among different smallholder fields, especially in orchard and vegetable systems. This study highlights that differences in farming practices within or among cropping systems should be taken into account when calculating nutrient balances and designing strategies of integrated nutrient management on a regional scale.

Keywords Nitrogen budget · Cereals · Orchards · Vegetables · Small-scale farms · Suburbs

Introduction

Agricultural production systems have been considered as a predominant aspect of natural resource consumption and environmental pollution on a global scale, including threshold countries such as China. Nowadays, China is the largest producer and consumer of mineral nitrogen (N) and phosphorus (P) fertilizers in the world (IFA 2007). Undoubtedly, the rapid development of fertilizer industry in China powerfully supported the recent achievement of Chinese agriculture, but also brought an unprecedented pressure on the limited and nonrenewable resources and fossil fuels due to the massive use of raw materials for fertilizer production (e.g. petroleum, coal, rock phosphate and sulfur minerals). The widespread overuse of mineral N fertilizer in China has caused considerable

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accumulation of reactive N in the soil, atmosphere and water bodies. Nitrogen has been a critical impact factor on the eutrophication of surface waters in China, which largely originated from crop production systems (Zhu and Chen 2002; Zhang et al. 2004) and intensive animal husbandry (Ju et al. 2005). From the 1960s to the 2000s, the contribution of N derived from cropping systems to major inland lakes and rivers in China markedly increased from 10 to 70 % of total N input from the combination of the cropping, livestock and peri-urban householder systems (Zhang et al. 2004). Nitrate contamination of groundwater in China is an important environmental issue, which is mainly driven by the excessive N application rates on farmland (Ju et al. 2004; Chen et al. 2005; Liu et al. 2005). A research performed in northern China reported that over 50 % of the locations investigated, which were distributed over an area of about 140,000 km², were characterized by nitrate contents in groundwater exceeding the EU drinking water standard of 50 mg nitrate (NO₃⁻) L⁻¹ (equivalent to 11.3 mg NO₃⁻-N L⁻¹) (Zhang et al. 1996). Therefore, in order to alleviate the serious environmental pollution, it is necessary to drastically reduce the loss of agriculture-derived N in China.

Nutrient balances are a useful tool for quantifying farmers' resources (e.g. mineral fertilizers, animal manure, crop products and residues) as well as environmental inputs (e.g. atmospheric deposition and biological N fixation) added into arable farming systems, and can provide well-founded information targeted at enhancing nutrient use efficiency and optimizing nutrient management (Oenema et al. 2003; Zingore et al. 2007). In addition, nutrient balances are usually adopted as indicators to estimate the potential risk of environment pollution (Schleef and Kleinhanss 1994; Oenema et al. 2003; Sun et al. 2008). For example, Oenema et al. (2005) found that in The Netherlands a 1 kg ha⁻¹ decrease of N surplus could reduce nitrate-N leaching to groundwater by 0.08 kg NO₃⁻-N ha⁻¹ and net N leaching to surface waters by 0.12 kg N ha⁻¹. Accordingly, the agriculturally environmental protection legislation and policy in most developed countries are generally established with an assistance of nutrient balance budgets (Brouwer 1998; Parris 1998; Oenema et al. 2003).

The urbanization rate in China dramatically increased from 17.9 to 46.6 % during 1978–2009 (NBSC, National Bureau of Statistics of China 2010), accompanied by a tremendous conversion of agricultural land to urban and

industrial area. In order to meet food requirements, the development of intensive farming was stimulated in the peri-urban areas of China. However, agricultural intensification in city suburbs has led to very high nutrient surpluses and potential environmental contamination with this trend still persisting for the future (Ju et al. 2006; Chen 2007). A case study at the provincial scale in Fujian Province of China indicated that two opposite trends existed in the topsoil-nutrient surplus reduction in cereal-based rural areas (annual surpluses of 107 kg N ha⁻¹ and 10 kg P ha⁻¹) but extreme nutrient overload in the peri-urban areas (annual surpluses of 1,000 kg N ha⁻¹ and 315 kg P ha⁻¹) (Cao et al. 2007). A similar trend was observed by Wei (2008), in which annual cropland N surplus markedly increased by nearly 300 kg N ha⁻¹ from 1995 to 2005 in peri-urban areas of Beijing, whereas by only 10 kg N ha⁻¹ on average in the adjacent rural area of Hebei Province. The rapid changes in cropping patterns in China have partly occurred as a result of the shifted structure of human diets with increasing demand for high-quality crop products (e.g. vegetables and fruits) as well as increased demand for animal products, which would be likely to alter farmers' resource allocation and nutrient balances between different cropping systems (Ju et al. 2006; Wang et al. 2007). However, an in-depth comparison of farming practices and nutrient balances among cropping systems in peri-urban areas of China is still lacking.

The objectives of the present study were to comprehensively estimate N inputs, outputs and balances of major cropping systems in a typical peri-urban area of China at a farmer's field scale and a cropping system scale, and to identify the predominant influencing factors of N balances, based on a 2-year survey. In addition, the potential risk of environmental pollution related to N surplus as well as improved N management options were also discussed in this study.

Materials and methods

The study area

This study was conducted in Shunyi District of Beijing Municipality (116°28'–116°58'E; 40°00'–40°18'N) with a total area of 102,100 ha, a typical district in the suburbs of Beijing. Shunyi District is located in the warm-temperate sub-humid zone featuring a continental monsoon climate with high temperatures and

high rainfall in the period of June to September. The mean annual temperature is 11.5 °C, with a monthly lowest mean temperature of 4.9 °C in January and a monthly highest mean temperature of 25.7 °C in July. The precipitation during the summer season (June to August) normally shares 75 % of the average annual rainfall of 500–700 mm, whilst scarce rainfall occurs during the winter season (December to February). The annual potential evapotranspiration amounts to 1,400 mm which is 2 times the amount of annual rainfall. Eutric and Luvic Cambisols (FAO 2006) with silty loam textures are predominant in Shunyi District as the primary Reference Soil Groups on farmland.

The population of Shunyi District in 2009 was 732,000, comprising 578,000 permanent and 154,000 temporary residents; thereby the population density (permanent and temporary residents) amounted to 718 persons km⁻². The urbanization rate in 2009 reached 47.2 %, showing a faster growth rate of 2.8 % per year (SBSD Statistical Bureau of Shunyi District 1999–2010) compared to the national growth rate of 1.2 % per year over the past decade (NBSC 1999–2010). Concomitantly, the cultivated land area in Shunyi District has shrunk to 31,031 ha in 2009, declining at a rate of 1,926 ha year⁻¹ (SBSD 1999–2010). The agricultural land use types mainly include cereals, orchards and vegetables, occupying about 62, 15 and 13 % of the total cultivated land area. Summer maize (*Zea mays* L.) and winter wheat (*Triticum aestivum* L.) are cultivated as major cereal crops and mostly in a traditional double-cropping rotation. Orchards mainly include pears (*Pyrus sorotina*), peaches (*Prunus persica*) and apples (*Malus pumila* Mill.), while a variety of vegetables under annual double or triple cropping are grown in Shunyi District. Smallholder farming is still the dominant agricultural production pattern here, despite the intensification of agriculture proceeding at an unprecedented rate as a result of the vicinity of the capital Beijing. Shunyi District has a very intensive animal production with very high livestock densities, amounting to 10.6 livestock units (LU) ha⁻¹ arable land in 2007 (calculated from SBSB 2008; 1 LU approx. 500 kg).

Farmer fields selection

Initial interviews with township and village officials were carried out in 2009 and 2010 to obtain general information on population, agricultural production

patterns, cropping systems, etc., ensuring the villages selected were representative of the peri-urban area of Beijing. Hence, three specific villages with cereals (maize and wheat double-crop rotation), orchards and vegetables (open-field vegetables) as the predominant cropping systems were identified. We collected information on farmers' resources and nutrient management practices from randomly selected smallholders, whose willingness to cooperate were also considered as a criterion for selection. A total of 68 households were monitored by filling in questionnaires during face to face interviews performed twice over a 2-year period (2008/2009 and 2009/2010), in combination with telephone inquiries for farmers' primary agricultural activities (e.g. fertilization, irrigation and harvest). For the farms investigated, agricultural production patterns could then be defined as smallholder farming patterns due to their farming characteristics such as the arable area ranging from 0.2 to 1.5 ha per farm (Table 1).

Calculation methods and data sources

Soil surface N balances were calculated separately on an annual and per hectare basis as well as on a farmer's field scale and a cropping system scale:

$$\text{Nitrogen balance} = (\text{IN1} + \text{IN2} + \text{IN3} + \text{IN4} + \text{IN5} + \text{IN6}) - \text{OUT1}(\text{OUT1a} + \text{OUT1b})$$

where IN1 and IN2 indicate N inputs from mineral fertilizer and animal manure, respectively. Nitrogen input from residues incorporated into the soil is expressed as IN3. Atmospheric N deposition (IN4), symbiotic N fixation (IN5) and N in irrigation water (IN6) are considered as N input components from natural environment. Nitrogen harvested from crop products (OUT1a) and crop residues (OUT1b) were calculated as N outputs through crop removal (OUT1).

Sources of data and parameters required and calculation methods for each of input and output items are summarized in Table 2. As origin of the data and parameters serving as inputs for the N budget, two categories can be distinguished: (1) farmers' surveys by way of face to face interviews accompanied with telephone inquiry, and (2) literature sources originated from extensive laboratory and field experiments that were mostly conducted in the local or adjacent region.

Table 1 Characteristics of smallholder farms investigated in three specific villages of Shunyi District, Beijing Municipality

Site	Number of farms surveyed	Arable area per farm (mean value and range) (ha)	Vegetation ^a	Crop rotations per year	Number of farms breeding animals	Animal types
Village 1	21	0.5 (0.2–0.9)	Maize, wheat	2	3	Pig, cattle
Village 2	24	1.0 (0.3–1.5)	Pear, peach	1	2	Pig, sheep
Village 3	23	0.4 (0.1–1.2)	Chinese cabbage, cabbage, cauliflower, cowpea, cucumber, eggplant, radish, scallion	2–3	0	–

^a Scientific names: maize: *Zea mays* L.; wheat: *Triticum aestivum* L.; pear: *Pyrus sorotina*; peach: *Prunus persica*; Chinese cabbage: *Brassica pekinensis* (Lour.) Rupr.; cabbage: *Brassica oleracea* L. var. *capitata* L.; cauliflower: *Brassica oleracea* var. *botrytis* L.; cowpea: *Pisum sativum* Linn.; cucumber: *Cucumis sativus* Linn.; eggplant: *Solanum melongena*; radish: *Raphanus sativus* L.; scallion: *Allium chinensis* G. Don

When selecting parameters from the references, several rules were adhered to ensure their suitability for this analysis. For calculating the cereal crop N removal (OUT1), the specific N contents of wheat and maize grains (2.14 and 1.30 % of dry weight) were calculated by the mean values from 11 typical experimental sites located in the Beijing region or in comparable adjacent agro-ecological zones. It should be noted that the variations in N contents of cereal grains (2.04–2.33 % for wheat; 1.21–1.48 % for maize) among different sites were relatively small, despite the application rates of fertilizer N varying from 150 to 400 kg N ha⁻¹ per crop (Lu et al. 1996b; Huang et al. 2000; Cai and Qin 2006; Li 2007). Several studies in the Beijing area indicated that crop N contents or uptakes remained constant or only slightly increased when N fertilizer application rates were higher than the optimum rates (Chen et al. 2002; Bai 2003; Zhao and Yu 2006). Our calculations of N removals by vegetables were based on site-specific experiments and literature investigations by Zhang et al. (2005), considering nutrient contents for different species of open-field vegetables in Beijing suburbs under local fertilization conditions. For orchards, N uptake data were derived from numerous experiments in the North China Plain (Jiang et al. 2007). For example, the N uptake by 100 kg fruit was estimated as 0.45 kg (0.42–0.47 kg) for pears and 0.50 kg (0.48–0.51 kg) for peaches.

The N application with animal manure (IN2) was quantified by the combination of farmers' surveys and reported data in the literature. The N content of animal manure originated from the National Agro-Technical

Extension and Service Centre (NATESC 1999a), in which animal categories and manure types (e.g. slurry, liquid and solid manure) were taken into account. Information on atmospheric dry and wet N deposition (IN4) was available for the study area and the mean annual N deposition rate was estimated at 77 kg ha⁻¹ year⁻¹ (Zhang et al. 2008; Shen et al. 2009). An asymbiotic N fixation (IN5) with a mean value of 10 kg N ha⁻¹ year⁻¹ was assumed for the investigated cropping systems where no leguminous crops are planted (Lu et al. 1996a). Nitrogen input with irrigation water (mainly derived from groundwater) was calculated on the basis of mean groundwater N concentrations measured in Shunyi District (Liu et al. 2001; Li et al. 2007) and the amounts of irrigation water applied (IN6).

Statistical analysis

The means of N inputs, outputs and balances among different cropping systems in a same year were compared by least-significant difference (LSD) method (equal variances assumed) and Tamhane's T2 method (equal variances not assumed) using SPSS 17 software package. The relationships between major N input/output components and N balances for each cropping system (0.95 confidence interval) were analyzed using the data analysis package of Microsoft Excel spreadsheets. Moreover, the two sample *t* test (0.95 confidence interval) was used to test significant differences in N inputs, outputs and balances between the 2 years surveyed within a same cropping system and performed with STATA 10.0 software package.

Table 2 Summary of calculation methods of N input and output items and sources of data and parameters required

Input/output items	Calculation methods	Data and parameters required	Sources
Mineral fertilizer (IN1)	$N \text{ content of mineral fertilizer} \times \text{Fertilizer application rate}$	Type and application rate of fertilizer N content of fertilizer	Farmers' surveys Farmers' surveys
Animal manure (IN2)	$N \text{ content of manure} \times \text{Amount of manure applied}$	Type and amount of manure applied N content of manure	Farmers' surveys [1] ^a
Incorporated residues (IN3)	$\text{Amount of residues (OUT1b)} \times \text{Portion incorporated}$	Amount of residues (OUT1b) Portion incorporated Amount of N returned to fields from residues ^b	OUT1b Farmers' surveys [2]
Atmospheric deposition (IN4)		Wet and dry deposition rates	[3,4]
Asymbiotic N fixation (IN5)		Crop types Asymbiotic N fixation rate	Farmers' surveys [5]
Irrigation water (IN6)	$N \text{ content of irrigation water} \times \text{Amount of irrigation water}$	Source and amount of irrigation water N content of irrigation water	Farmers' surveys [6,7]
Crop product (OUT1a)	$N \text{ content of crop product} \times \text{Yield}$	Type and yield of crop product N content of crop product	Farmers' surveys [8–12]
Residues (OUT1b)	$\text{Yield of crop product} \times \text{Residues/grain ratio} \times N \text{ content of residues}$	Type and yield of crop product Residue/grain ratio N content of residues	Farmers' surveys [12, 13] [1, 12]
Crop removal ^c (OUT1)	Method 1: $\text{Sum of OUT1a and OUT1b}$ Method 2: $\text{Yield} \times N \text{ uptake by crop}$	Type and yield of crop product N uptake by crop	Farmers' surveys [2]

^a References: [1] NATESC (1999a); [2] Jiang et al. (2007); [3] Zhang et al. (2008); [4] Shen et al. (2009); [5] Lu et al. (1996a); [6] Liu et al. (2001); [7] Li et al. (2007); [8] Cai and Qin (2006); [9] Huang et al. (2000); [10] Li (2007); [11] Lu et al. (1996b); [12] Zhang et al. (2005); [13] NATESC (1999b)

^b Nitrogen added to the soil by the fallen leaves in orchards was considered as N input from incorporated residues

^c Amounts of N removed by crops (OUT1) were calculated by using Method 1 for cereals and vegetables; Method 2 for fruit trees

Results

Annual nitrogen balances of three major cropping systems

The mean annual N fluxes regarding N input, output and surplus/deficit in 2008/2009 and 2009/2010 for cereal, orchard and vegetable cropping systems in Shunyi District are shown in Table 3.

Nitrogen fertilizers (mineral fertilizer and animal manure) were the largest contributors of total N inputs,

accounting for 76, 83 and 95 % of total N inputs in cereal, orchard and vegetable cropping systems, respectively. However, there were considerable differences in the relative contribution of mineral fertilizer and animal manure to total N inputs. For instance, about 74 % of the total N entering the cereal system was derived from mineral fertilizer, or nearly 42 times as much as that from animal manure. In contrast, in the vegetable system 65 % of the total N inputs were from animal manure, while only 30 % originated from mineral fertilizer. Annual N inputs with mineral fertilizers to

Table 3 Nitrogen inputs, outputs and balances in the cereal, orchard and vegetable cropping systems of Shunyi District, Beijing Municipality for 2008/2009 and 2009/2010 (kg N ha⁻¹ year⁻¹)

Categories of N balance	Cropping systems						Mean value	Mean value
	Cereals (n = 21)		Orchards (n = 24)		Vegetables (n = 23)			
	2008/2009	2009/2010	2008/2009	2009/2010	2008/2009	2009/2010		
<i>Inputs</i>								
Mineral fertilizer	599 a (303–919) ^a	600 a (395–798)	600	349 b (0–794)	358	638 a (216–935)	701 a (216–1,043)	669
Animal manure	14 c (0–144)	14 c (0–144)	14	209 b (0–1,808)	178	1,434 a (370–3,448)	1,451 a (393–3,448)	1,443
Incorporated residues	108	102	105	18	18	0	0	0
Atmospheric deposition	77	77	77	77	77	77	77	77
Asymbiotic N fixation	10	10	10	10	10	10	10	10
Irrigation water	4	5	4	6	6	14	14	14
Total input	812 (510–1,076)	808 (582–978)	810	667 (120–2,241)	645	2,173 (684–4,414)	2,252 (1,099–4,345)	2,213
<i>Outputs</i>								
Crop removal (products + residues)	286 b (187–324)	272 b (201–313)	279	130 c (34–225)	126	645 a (349–1,101)	685 a (299–987)	665
<i>Balances</i>								
N surplus/deficit	526 b (213–890)	536 b (320–743)	531	537 b (–49–2,054)	519	1,527 a (334–3,555)	1,568 a (732–3,478)	1,548

Within a same year investigated, mineral fertilizer, animal manure, crop removal and N surplus/deficit were compared among different cropping systems (different letters indicate significant differences at 0.05 level)

^a The range of values

^b The mean value of the 2 years investigated

cereal and vegetable cropping systems were significantly higher than those to the orchard system ($P < 0.05$), but there was no significant difference between the two former systems ($P > 0.05$). The annual N input from animal manure was by far the largest in the vegetable system, amounting to $1,443 \text{ kg N ha}^{-1} \text{ year}^{-1}$, followed by the orchard system and then by the cereal system ($P < 0.05$).

The amount of N removed by crop harvest was particularly different among the three cropping systems ($P < 0.05$), in an order of vegetable > cereal > orchard systems.

High N surpluses occurred across all three cropping systems in Shunyi District over both years, especially in the vegetable system where the mean annual N surplus reached $1,548 \text{ kg N ha}^{-1} \text{ year}^{-1}$. There were no significant differences in N surpluses between cereal and orchard systems ($P > 0.05$). This was because total N additions to orchards were lower

compared to cereal fields, but at the same time N removal by orchards was also smaller.

Comparison of N input and output data and N balances between the 2 years monitored (2008/2009 and 2009/2010) did not yield any statistically significant differences between years for each of the three cropping systems ($P > 0.05$).

Variation of nitrogen balances among farmers' fields

Mineral N fertilizer application rates in the range from 451 to $750 \text{ kg N ha}^{-1} \text{ year}^{-1}$ were mainly concentrated on 76 % of the farms within the cereal system. But larger differences in mineral N fertilizer inputs among farms were observed in the orchard and the vegetable systems. In the orchard system, mineral fertilizer application rates ranging from 1 to 150, 151–300 and 301–450 $\text{kg N ha}^{-1} \text{ year}^{-1}$ were

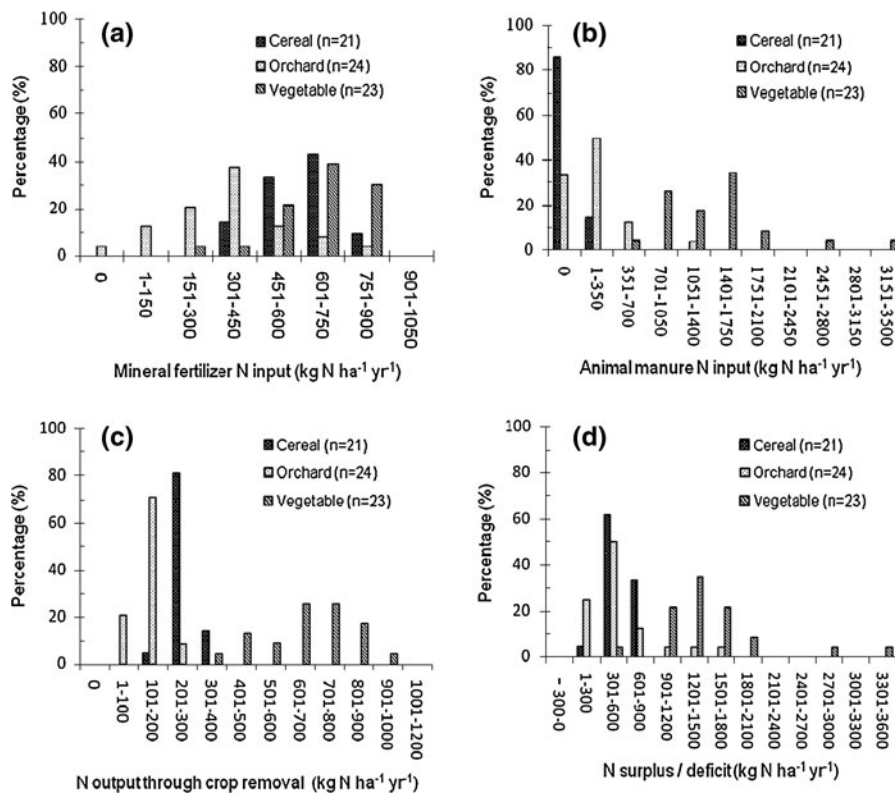


Fig. 1 Frequency distribution of number of surveyed farms based on different classifications of mineral fertilizer N input (a), animal manure N input (b), N output through crop removal

(c) and N balance (d) (mean value of the investigated 2 years) for the three cropping systems

distributed among 13, 21 and 38 % of the orchards, respectively. Within the vegetable system, 22, 39 and 30 % of the farmers applied mineral N at rates ranging from 451 to 600, 601–750 and 751–900 kg N ha⁻¹ year⁻¹ (Fig. 1a). It was evident that 86 % of farmers did not apply any animal manure to cereal fields. In orchards, animal manure application was more common (>60 % of farmers) compared to the cereal system. However, animal manures were applied in all of the vegetable farms, with a large range of N application rates from 369 to 3,448 kg N ha⁻¹ year⁻¹ (Fig. 1b).

Nitrogen removed by crop harvests at more than 70 % of investigated farms were concentrated in narrower ranges between 101 and 200 kg N ha⁻¹ year⁻¹ for the orchard system, and between 201 and 300 kg N ha⁻¹ year⁻¹ for the cereal system, contrasting with a greater variation among different vegetable farms (Fig. 1c).

Consequently, the N surpluses and deficits within each cropping system (or within each village) also showed an uneven distribution, resulting from the variability of fertilizer application and crop removal among different farms (Fig. 1d). For instance, the N surplus varied greatly between 901 and 2,100 kg N ha⁻¹ year⁻¹ on nearly 80 % of vegetable-growing farms.

Impact of fertilization and crop production on nitrogen surplus

Relationships between fertilizer N inputs (including mineral fertilizer and animal manure) and N outputs through crop removal versus N surpluses/deficits were analyzed for all three cropping systems. Fertilizer N application was strongly and positively correlated with soil N balances ($P < 0.05$, Fig. 2a–c), whereas no declining trends in N surplus were observed with increasing N output through crop removal ($P > 0.05$, Fig. 2d–f).

The N surplus in the cereal system was particularly due to excessive mineral N fertilizer application ($P < 0.05$, Fig. 3a), but irrespective of animal manure ($P > 0.05$, Fig. 3d). In contrast, the N surplus in the vegetable system was strongly and positively correlated with animal manure ($P < 0.05$, Fig. 3f), but there was no significant correlation with mineral fertilizer ($P > 0.05$, Fig. 3c). In the orchard system, both mineral fertilizer and animal manure were significantly positive contributors to the N surplus ($P < 0.05$, Fig. 3b, e).

Discussion

Nitrogen balances related to cropping systems

The current study showed that the mean annual N surpluses ranged from 519 kg N ha⁻¹ year⁻¹ in the orchard and 513 kg N ha⁻¹ year⁻¹ in the cereal system to 1,548 kg N ha⁻¹ year⁻¹ in the vegetable system. Besides the N losses to the environment, N accumulation in soils is also a possible fate of surplus N, especially under the sub-humid climate. A parallel study conducted by L. Heimann et al. (unpublished) on 14 plots under the same three cropping systems in Shunyi District over a 2-year period (2009–2010) has found soil mineral N (NO₃⁻-N + NH₄⁺-N) contents in a range of 500–1,000 kg N ha⁻¹ in 0–2 m soil depth (Table 4). The soils of the villages in Shunyi District investigated in this study and in that by Heimann et al. are similar in texture (silty loam) and thus have a similarly high water holding and mineral N storage capacity. The nitrate storage and leaching characteristics are comparable in both studies, and, consequently, comparisons between the N balance surpluses in this study and the soil mineral N contents found by Heimann et al. may be drawn. These mineral N contents can also partly validate the differences in N balance surpluses among the three cropping systems, especially for the 0–20 cm soil depth increment.

A wide-sampling survey of soil nitrate contents in Beijing suburbs was conducted by Du et al. (2009). They reported that the average NO₃⁻-N content in the 0–30 cm soil profiles of vegetable fields was 46.2 mg kg⁻¹ (194 kg N ha⁻¹) and significantly greater than those in cereal farmlands of 11.9 mg kg⁻¹ (50 kg N ha⁻¹) and orchards of 14.4 mg kg⁻¹ (60 kg N ha⁻¹). The vegetable production in peri-urban areas has been characterized by an extremely high soil surface N surplus. For instance, a study conducted in two representative peri-urban areas (Nanjing and Wuxi) of southeastern China reported that the mean annual N surpluses of vegetable farmlands in the two sites amounted to 954 and 1,237 kg N ha⁻¹ year⁻¹, respectively (Wang et al. 2008). A high N surplus up to 882 kg N ha⁻¹ year⁻¹ in small-scale vegetable farming systems near Hanoi (Vietnam) was also presented by Khai et al. (2007). The N balances of greenhouse vegetable systems were not estimated in the current study. But it has been shown that N surpluses in such systems can reach 3,327 kg N ha⁻¹ year⁻¹ in Huimin

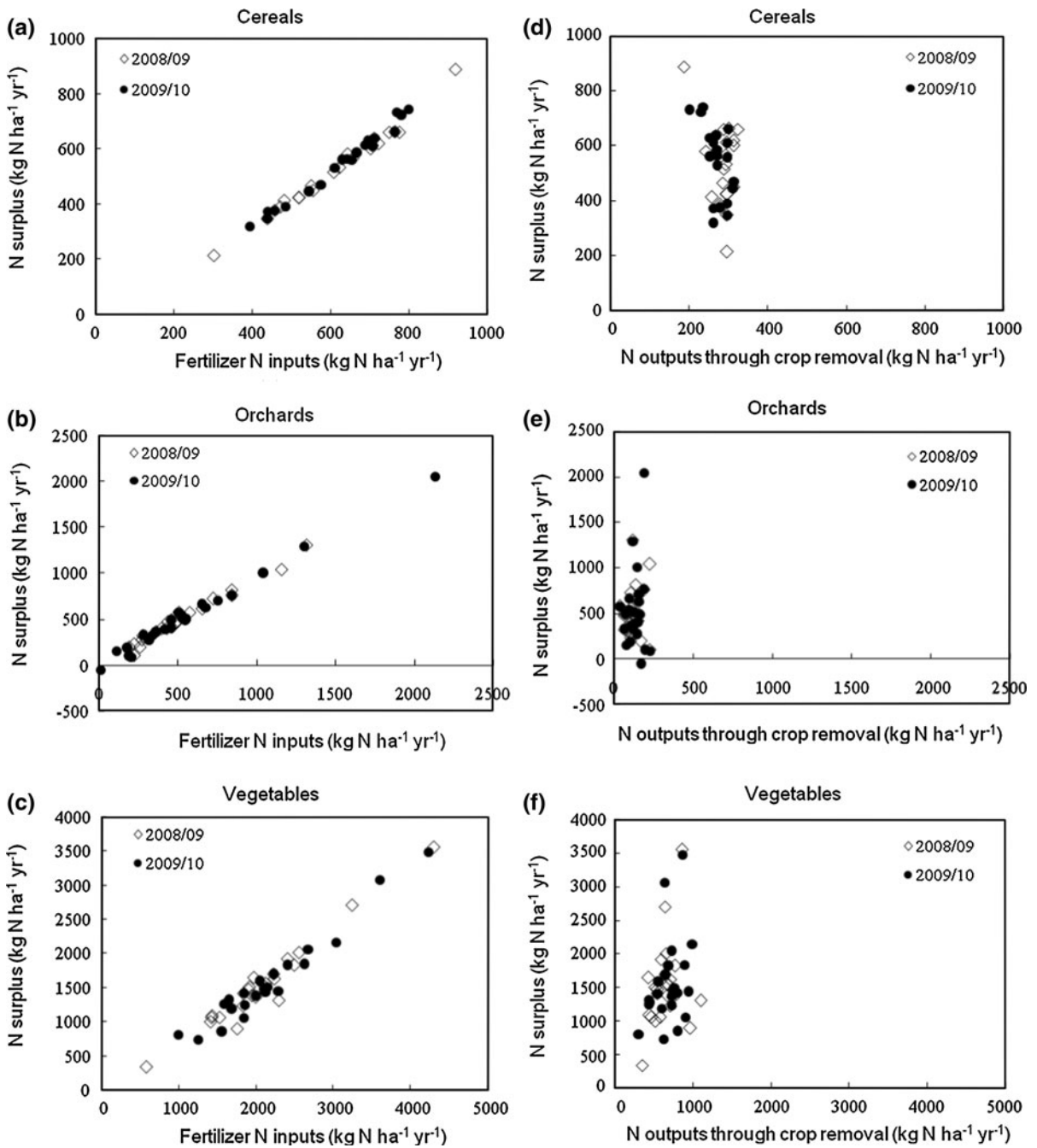


Fig. 2 Relationship of fertilizer N inputs (including mineral fertilizer and animal manure) (a–c) and N outputs through crop removal (d–f) versus N balance surplus for three major cropping systems of Shunyi District, Beijing in 2008/2009 and 2009/2010

City, Shandong Province of China (Ju et al. 2006), much higher than those in open-field vegetable systems.

Two contrasting trends in the use of animal manure were found in our study: On the one hand, there was an excessive use in the vegetable cropping system and on

the other hand, an insufficient use in the cereal cropping system. Animal manure has been ascertained to be the major nutrient source of vegetable production in Beijing region, contributing to over 50 % of the N input, 60 % of the P input and 90 % of the K input (Chen et al.

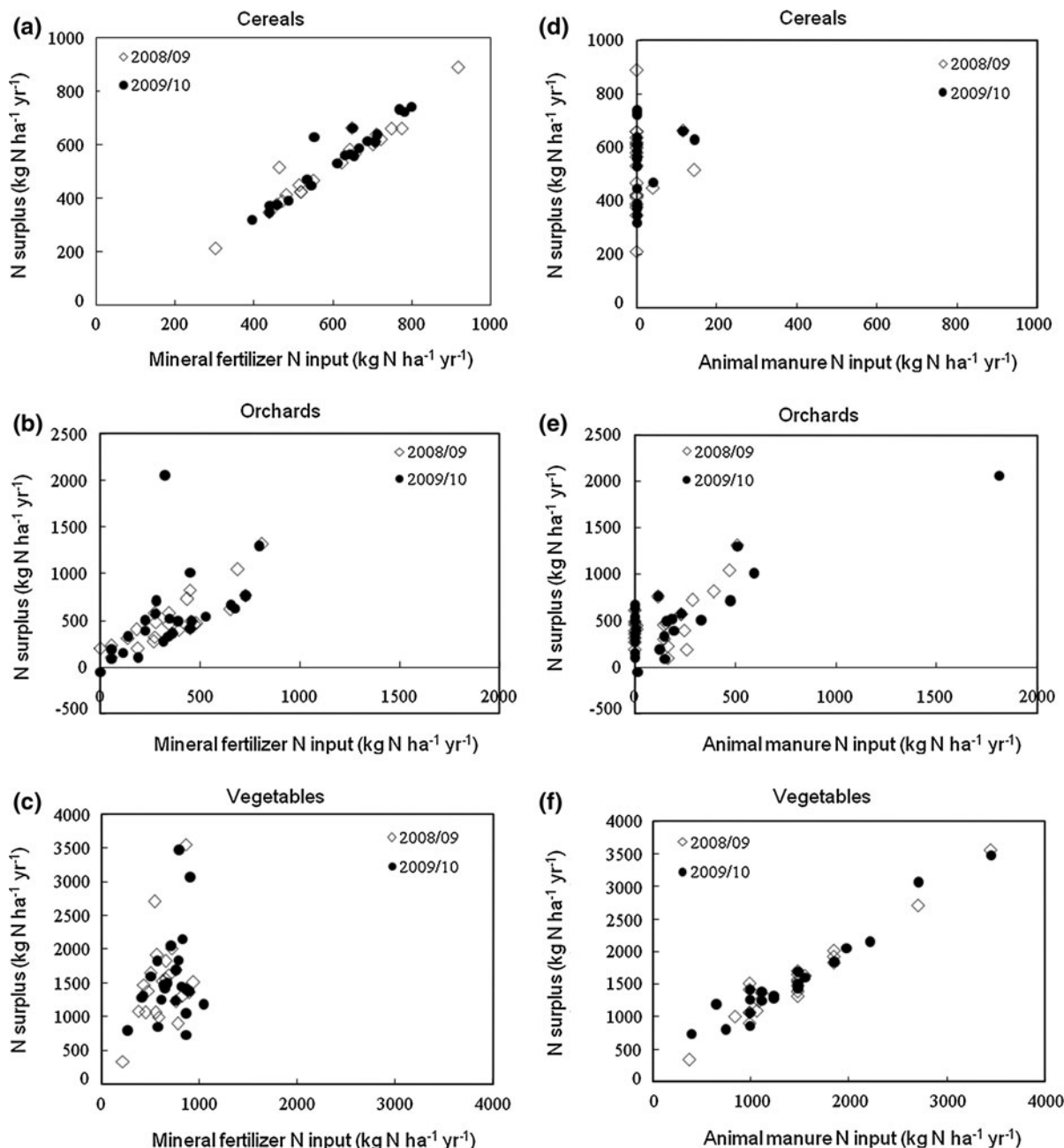


Fig. 3 Relationship of mineral fertilizer N input (a–c) and animal manure N input (d–f) versus N balance surplus for three major cropping systems of Shunyi District, Beijing in 2008/2009 and 2009/2010

2004). Our study has shown that the average amount of applied N to the vegetable system via animal manure reached 1,443 kg N ha⁻¹ year⁻¹. This was significantly higher than that in most European countries, where ranges were from 43 to 307 kg N ha⁻¹ year⁻¹ in intensive livestock production systems (Oenema 2004).

These also exceeded the European Nitrate Directive threshold of 170 kg N ha⁻¹ for manure N application in the EU (*ibid.*). In contrast, manure input was not important in the cereal cropping system (14 kg N ha⁻¹ year⁻¹), as a result of the low economic benefits from cereal production, compared to high production costs

Table 4 Mean mineral nitrogen (NO_3^- -N + NH_4^+ -N) contents (kg N ha^{-1}) in 0–20 cm and 0–200 cm soil profiles of 14 plots under three typical cropping systems of Shunyi District, Beijing during 2009–2010 (L. Heimann et al., unpublished)

Cropping systems	Soil profile depth (cm)	Date of sampling							
		March 2009		October 2009		March 2010		October 2010	
		Mean \pm SD	<i>n</i>	Mean \pm SD	<i>n</i>	Mean \pm SD	<i>n</i>	Mean \pm SD	<i>n</i>
Wheat/maize	0–20	70 \pm 37	8	39 \pm 32	8	133 \pm 136	6	52 \pm 26	6
Orchards	0–20	77 \pm 39	3	38 \pm 7	3	164 \pm 208	3	62 \pm 18	3
Vegetables	0–20	105 \pm 61	3	170 \pm 201	3	140 \pm 37	2	51 \pm 5	2
Wheat/maize	0–200	492 \pm 367	8	453 \pm 269	8	627 \pm 470	6	1,049 \pm 561	6
Orchards	0–200	852 \pm 487	3	205 \pm 342	3	746 \pm 367	3	1,313 \pm 600	3
Vegetables	0–200	536 \pm 307	3	883 \pm 777	3	1,033 \pm 34	2	720 \pm 158	2

n number of plots. Bulk densities used for conversion on an area basis: Actual bulk densities determined for the 0–20 and 20–60 cm depth increments; mean bulk density of 1.4 kg dm^{-3} used for the depth increments in 60–200 cm

and labor inputs when applying animal manure. This has led to an inefficient recycling of organic residues from animal production in the study area, which caused serious environmental problems (Ma et al. 2009). Investigating farmers' perception of fertilizer application, we found that farmers applied animal manure mainly based on their own experiences and without scientific guidance. They mainly lacked recognition of the nutrient contents in animal manure. Therefore, the nutrient demand-oriented use of organic resources and the optimized reallocation of animal manure among cropping systems are the primary tasks to solve the issue, besides a reduction in livestock densities.

Potential risk of environmental pollution

Nitrogen fertilizer use in China has largely contributed to the achievement of self-sufficiency in food production and the improvement of soil quality since the early 1970s, yet excessive N fertilization during the past two decades has resulted in serious environmental problems because of the accumulation of reactive N from agricultural origin in soil, water and atmosphere (Zhu and Chen 2002; Ma et al. 2010). Nitrogen balance budgets offer individual farmers, extension workers and government a practical and useful agri-environmental tool or indicator (by using a reference level) to monitor and assess changes in N surpluses. Schleef and Kleinhanss (1994) proposed that 100 kg ha^{-1} of annual N surplus could be taken as a baseline for possible nitrate leaching into ground and surface waters on a regional/national scale. Besides, to weigh agronomic-economic versus environmental

consequences, some studies suggested that reference values for annual cropland N surpluses could be set at 60 kg N ha^{-1} on sandy soils and 100 kg N ha^{-1} on clayey soils (Oenema et al. 2003). Based on a systematical assessment of surface N balances in crop production systems in China, Sun et al. (2008) identified that the potential and high risk regions were associated with N balance surpluses of more than 100 and 180 kg N ha^{-1} , respectively. Therefore, the annual N surpluses (mostly more than $500 \text{ kg N ha}^{-1} \text{ year}^{-1}$) of all three cropping systems in Shunyi District have obviously gone far beyond the above-mentioned "thresholds" of N surplus, suggesting a great potential risk of environmental contamination.

As to the N losses from these cropping systems, according to Ju et al. (2009), 22–40 % of the N applied in wheat–maize rotation system on the North China Plain could be emitted via ammonia (NH_3) volatilization, nitrate leaching and denitrification. Generally, it is difficult to measure all N losses via different pathways simultaneously, so frequently, indirect estimates are employed as well. A study conducted by Cai et al. (2002) in the same area showed that approximately 39 % (range: 10–67 %, calculated as the difference between N surplus and soil N storage) of the total N applied could apparently be released to the environment. If the total N losses in the current study were assumed to be 39 % of the total N input in the cereal system, they would amount to $316 \text{ kg N ha}^{-1} \text{ year}^{-1}$. Liang (2011) summarized 272 experiments of 178 published papers in the past three decades and showed that on average 35 % of total N inputs in the open-field vegetable system were emitted into atmosphere and

water bodies. Thus the estimated annual N loss of the vegetable system in the study area probably reached $781 \text{ kg N ha}^{-1} \text{ year}^{-1}$. A clear qualitative relationship between N surplus and nitrate concentrations in the groundwater in a given crop-growing area has been defined, meaning that massive N accumulated in fields is likely to increase the risk of groundwater nitrate contamination (Ju et al. 2006). Several studies conducted in Shunyi District, in other parts of the Beijing region and in northern China have shown that leaching is the dominant pathway of N loss in the intensive vegetable production areas, also due to their frequent irrigation. Furthermore it has been found that nitrate contents in the groundwater of vegetable-producing areas in most cases by far exceeded the EU drinking water standard of $50 \text{ mg NO}_3^- \text{ L}^{-1}$, and were distinctly higher than those in other cropping systems (Zhang et al. 1996; Liu et al. 2001; Du et al. 2009). Although a relatively lower absolute amount of N loss was assessed for the cereal system (occupying 62 % of total cultivated area) compared to the corresponding value for the vegetable system (occupying 13 % of total cultivated area), the potentially negative impact of the predominant winter wheat-summer maize double-cropping system, characterized by a long-term N accumulation, on the environment is also non-negligible.

Strategies for optimizing nitrogen management

There is a great potential to reduce N input for the three cropping systems through optimized nutrient management. Balanced N fertilization is an effective measure to recommend fertilizer N application rates and decrease N losses from fertilizers (Oenema et al. 2009; Velthof et al. 2009; Ma 2010). Therefore, according to Velthof et al. (2009) and Ma (2010), this method was adopted here to evaluate the reduction potential of N fertilization. Taking into account the crop N uptake capacity, the appropriate N fertilization rates were estimated to be $307 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for cereals, $158 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for orchards and $831 \text{ kg N ha}^{-1} \text{ year}^{-1}$ for vegetables. When unavoidable N losses of agricultural inputs are considered, consequently, the recommendable reductions in fertilizer N (mineral fertilizer + animal manure) would be 297, 299 and 591 kg N ha^{-1} , or 48, 56 and 28 % of the current fertilizer N inputs in the cereal, orchard and vegetable cropping systems, respectively. This is in agreement with Ju et al. (2009), stating that 30–60 % of the current

amounts of N applied in intensive agricultural systems in China could be saved by knowledge-based N management, without losses in yield or grain quality. However, an obvious decline in fertilizer N use is unlikely to be achieved in the short term, partly because of a lack of functioning extension services unable to promote agricultural technologies for farmers. Another limitation is the general unwillingness by farmers to accept any reductions in N fertilizer application rates, despite stable crop yields when less N fertilizers are applied, based on the currently recommended “integrated nutrient management”.

The present study has shown that mineral N fertilizer and manure N application rates, crop N removal and N balance surpluses among smallholders’ fields had large variations both within and between cropping systems, and particularly, in the orchard and vegetable systems (Fig. 1). At a farmer’s field scale, on only 20 % of the orchards and vegetable fields surveyed were the amounts of applied N equal to or lower than the recommendations of balanced N fertilization. As illustrated in Fig. 4, there were no obvious differences in N surpluses for each farm investigated in the three cropping systems between 2008/2009 and 2009/2010. It can partly validate that, under the current “experience-based” fertilization, a persistent high N surplus situation would persist in the study area in the short term. Consequently, an efficient recommendation system for fertilization suitable to the smallholder’s field scale (still prevailing in China) is urgently required.

Conclusion

Positive soil surface N balances resulted across all three cropping systems in the Shunyi District of Beijing over the 2-year period (2008/2009 and 2009/2010) investigated. The annual N surplus in the open-field vegetable system was significantly higher than that in wheat–maize rotation and orchard systems, respectively. Chemical fertilizer N input for the cereal system, animal manure N input for the vegetable system and both of them for orchards were detected as major sources of N surpluses. As a consequence, a new strategy for optimizing nutrient management, especially a reduction in mineral N fertilizer application rates in particular on the plots receiving animal manure, a generally more effective utilization of animal manure

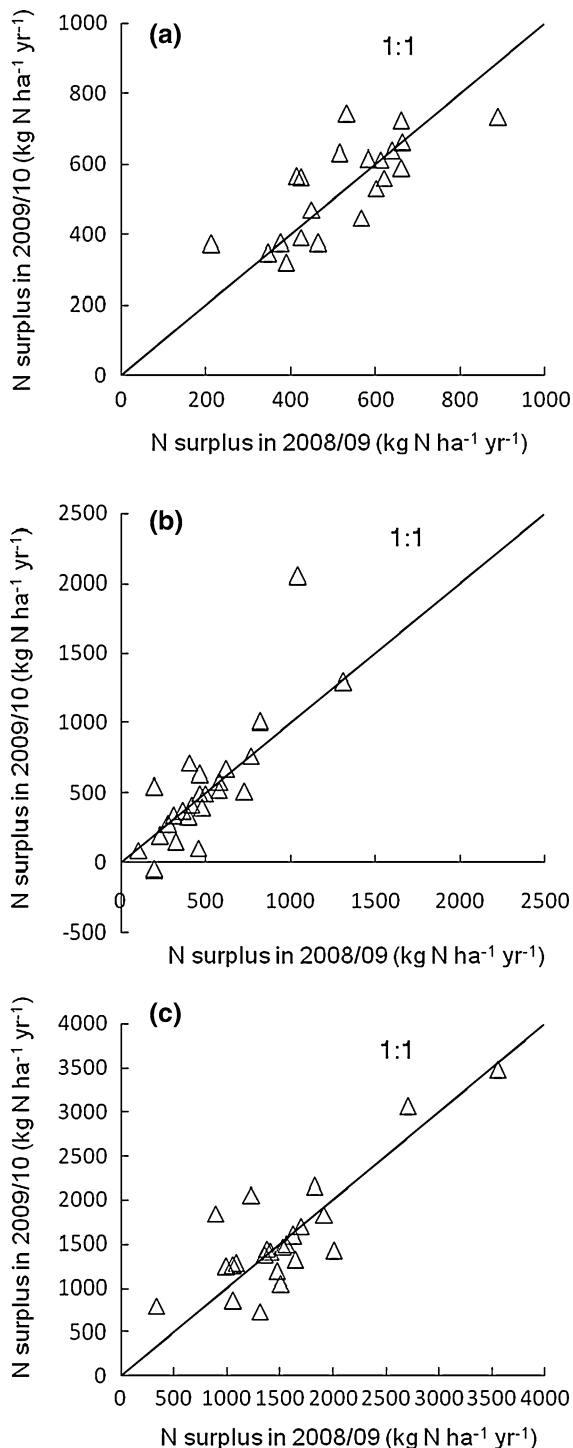


Fig. 4 Comparison of N balance surpluses for each smallholder's field investigated between 2008/2009 and 2009/2010 for the cereal (a), orchard (b) and vegetable (c) cropping systems of Shunyi District, Beijing Municipality

and balanced fertilization should be carried out and extended to the farmers. Furthermore, a large variability in the allocation of farmers' resources and farming practices was found between the three cropping systems as well as within each of them. Such variability should be taken into account when calculating nutrient balances and designing options to increase resource use efficiency and alleviate environmental pollution on a regional scale. Moreover, by specifically targeting those farmers at the upper end of the range of fertilizer application rates in each village, it is possible to reduce the average N surplus in each village quite effectively.

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