Nutr Cycl Agroecosyst
DOI 10.1007/s10705-015-9737-y

PERSPECTIVE

Changes in phosphorus use and losses in the food chain of China during 1950–2010 and forecasts for 2030

Zhaohai Bai · Lin Ma · Wenqi Ma · Wei Qin · Gerard L. Velthof · Oene Oenema · Fusuo Zhang

Received: 3 February 2015 / Accepted: 23 September 2015

Abstract China has become the largest mineral phosphorus (P) fertilizer consumer in the world, but current use is not sustainable. Here, we report on a quantitative analysis of the P use and losses in the food production–consumption chain and of their relationships with socio-economic indicators for the years 1950–2010. Pathways to a more sustainable P use in 2030 were explored through scenario analyses, using the Nutrient flows in Food chains, Environment and Resource use model. Non-linear relationships were observed between changes in P use and changes in gross domestic production (GDP), suggesting a decoupling of P use from the main economic driver. More or less linear relationships were observed between changes in P use and changes in the percentages of vegetable and fruit and animal derived food in human diets. Total P losses increased from 0.2 Tg in 1950 to 3.1 Tg in 2010, while P use efficiency in the food chain decreased from 35 % in 1950 to 6 % in 2010. Our estimates suggest that 79 Tg P has accumulated in agricultural soils, 16 Tg P accumulated in landfill, and 48 Tg P has leached or has been discharged to water bodies during the past 60 years. Most of the accumulation and discharges took place in the last 10 years. We analyzed five options for increasing P use efficiency in the food chain by 2030, i.e., balanced P fertilization in crop production, precision animal P feeding, improved manure management, diet changes, and the integration of these four options. The integral adoption of these four options will increase P use efficiency in the food chain from 6 % in 2010 to 26 % in 2030. Total mineral P fertilizer use will decrease by 69 % and P losses by 68 % relative to the business as usual scenario. In conclusion, current P fertilizer use and losses are coupled to dietary choices, but have become

Electronic supplementary material The online version of this article (doi:10.1007/s10705-015-9737-y) contains supplementary material, which is available to authorized users.

Z. Bai · L. Ma (✉)
Key Laboratory of Agricultural Water Resources, Center for Agricultural Resources Research, Institute of Genetic and Developmental Biology, The Chinese Academy of Sciences, 286 Huaizhong Road, Shijiazhuang 050021, Hebei, People's Republic of China
e-mail: malin1979@sjziam.ac.cn

W. Ma
College of Resources and Environmental Sciences, Agricultural University of Hebei, Baoding 071001, People's Republic of China

W. Qin · O. Oenema
Department of Soil Quality, Wageningen University, P.O. Box 47, 6700 AA Wageningen, The Netherlands

G. L. Velthof · O. Oenema
Alterra, Wageningen University and Research Centre, P.O. Box 47, 6700 AA Wageningen, The Netherlands

F. Zhang
College of Resources and Environmental Sciences, China Agriculture University, Beijing 100193, People's Republic of China

Published online: 01 October 2015

decoupled from GDP. Further decoupling may occur when P use is defined by science-based P requirements for crops, animals and humans.

Keywords Scenario analysis · Fertilizer · Losses · Crop · Animal · Socio-economic

Introduction

Phosphorus (P) is an essential nutrient, needed for cellular function, reproduction (DNA, RNA) and plant development (Richardson 2009; Smil 2000). Soils usually do not contain sufficient plant-available P for economic optimal crop production and cropland is therefore fertilized. Without applications of P fertilizer, crop yields would be limited and it would be difficult to produce enough food for the projected 9 billion people in 2050 and thereafter (UN 2010). Globally, 80–90 % of the mined P is used as mineral P fertilizer in food production. However, only about 20 % of the fertilizer P is taken up by the crop and ends up in food consumed by people (Cordell et al. 2009; Syers et al. 2008). The remainder accumulates in agricultural soil, or is lost into the wider environment at various stages of the food production–consumption chain. Losses of P to surface waters contribute to eutrophication and biodiversity loss in rivers, lakes and coastal waters (Carpenter 2008; Carstensen et al. 2014; Conley et al. 2009). Hence, there is urgent need to improve P use efficiency and to protect the environment at the same time (Godfray et al. 2010; Withers et al. 2015).

China plays a key role in global P fertilizer use, as it was the largest mineral P fertilizer consumer (30 %) in the world in 2010 (FAO 2014). The large P use has resulted in a relatively low P use efficiency compared to many other regions in the world (MacDonald et al. 2011). The increased P use in China has been ascribed to the increased food and feed needs by the increasing human and animal populations (Ma et al. 2012a, b). The use has been facilitated also by the booming economy, the existence of native P rock reserves, and governmental subsidies to the mineral P fertilizer production and distribution sectors and farmers (Zhang et al. 2008; Li et al. 2013). The rapidly increasing urban population has increased the consumption of animal derived foods from about the 1990s, which has increased the per capita P footprint (Metson et al. 2012). The increased consumption of vegetables and fruits after 1990s has contributed to an increase of P fertilizer use, because of the relatively large P fertilization rate in vegetable and fruit production (Yan et al. 2013).

Changes in human diets are often related to changes in gross domestic production (GDP) and urbanization. An increasing consumption of animal protein has been observed in many developing countries with an increasing GDP (Tilman and Clark 2014). However, P fertilizer use, P losses and P use efficiencies in the food chain have not been related quantitatively to socio-economic developments yet. Linking P use and P use efficiency to changes in GDP and to the consumption of animal derived food, vegetables and fruits may help to identify (de)coupling mechanisms. Insights in such mechanisms may help to improve forecasts of future mineral P fertilizer use and losses, especially when supported with scenario analyses. Improved forecasts and scenario analysis may subsequently guide policy and management decisions.

Forecasts indicate that the human population in China will peak at about 1.4 billion in 2030 (FAO 2014). The ratio of urban to rural populations will increase from 1:1 in 2010 to 2:1 in 2030 (FAO 2014). These changes greatly increase the demand for animal-derived food, vegetables and fruits, because urban people tend to consume more animal protein, vegetables and fruits than rural people. Evidently, increasing the production of food and increasing the P use efficiency simultaneously represents an enormous challenge for China.

Here, we report on a quantitative analysis of P use, P losses and P use efficiencies in the food chain of China for the period 1950–2030, and thereby extend and recast previous estimates. The objectives of the study were (1) to quantitatively relate P use, P losses and P use efficiencies to socio-economic drivers, through an updated version of the Nutrient flows in Food chains, Environment and Resource use (NUFER) model; and (2) to examine the P use, P losses and P use efficiencies for the year 2030 for five scenarios.

Materials and methods

Data sources

Statistical sources (MOA 2009, 2013; NBSC 2009, 2013) were used for records about changes in GDP, human population, urbanization rate, fertilizer use, crop yields, cultivated areas, and number of animals, for the period 1950–2010. Average food consumption per capita, total food import and export, production and use of green manure, fodder and natural grass, and vegetables and fruits were derived from available statistical data for the years 1980–2010. For the period 1950–1980, we assumed that some data were the same as in 1980, for example average food consumption rate, as no statistical data were available (See Tables S1-S3, Figure S1).

Description of the NUFER model

Nutrient Flows in Food Chain, Environment and Resource use is a deterministic and static model that calculates P inputs and outputs in crop and animal production and food processing, retail and consumption, at the regional scale in China on an annual basis (Ma et al. 2010). The food chain in NUFER is perceived as a "pyramid" with four main compartments: (1) crop production (including the rootable soil layer), (2) animal production (including managed aquaculture), (3) food processing, and (4) households (Fig. 1). NUFER consists of an input module with activity data and transformation and partitioning coefficients, a calculation module with equations, an optimization module, and an output module. NUFER allows assessment of the P flows in the pyramid in two directions, viz., from the food production side and from the consumption side (Ma et al. 2010). For detailed information about the P inputs and outputs of each sector, and about the calculation methods we refer to the supplementary information (SI).

Three main accumulation pools/sinks were distinguished, namely agricultural land, non-agriculture land (landfill plus farm yards) and water bodies. The P accumulation in agriculture land was considered to occur in the food production and consumption system, while the P accumulation in non-agriculture land and P losses to water bodies were considered as losses from the food system (outside the system boundary; see Fig. 1). The accumulation of P in agricultural soils was estimated from soil P balances. The accumulation of P in water bodies was estimated from the P losses from crop and animal production, food processing sector and households via loss pathways-specific (surface runoff, leaching, erosion and discharges) coefficients. The accumulation of P in landfill plus farmyards (including lagoons and road sides) was estimated from discharge-specific loss fractions for manures, residues and wastes from animal production, food processing and household sectors. The accumulation of manure P in animal confinements (i.e., the fraction not collected) was considered to be part of the accumulation of P in landfill plus farmyards. The partitioning of the manure P accumulation between water bodies and landfill plus farmyards is not well-

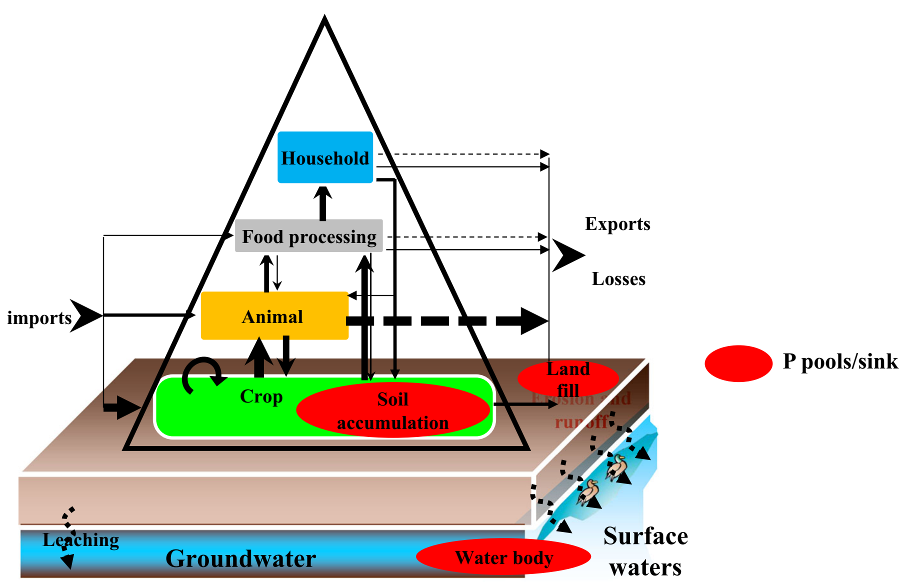


Fig. 1 Representation of the food production–consumption chain and its system boundaries, flows and pools in the NUFER model (after Ma et al. 2010). *P* phosphorus

known; in this study we assumed that the ratio between manure P discharges to water bodies and to landfill plus farmyards was 4:1. For more information about the NUFER model see supplementary information.

Calculation of phosphorus use efficiency

The P use efficiency in crop production (PUEc) was defined as (see Ma et al. 2010, 2012b):

$$\mathrm{PUEc} = \left(\mathrm{Oc}_{\text{main products}}/\mathrm{Ic}_{\text{total}}\right) \times 100\,\% \tag{1}$$

where, $\mathrm{Oc}_{\text{main products}}$ is total P output via harvested main crop products (excluding the residues), and $\mathrm{Ic}_{\text{total}}$ is the total P input to crop production. The P use efficiency in animal production (PUEa) was defined as:

$$\mathrm{PUEa} = \left[\left(\mathrm{Oa}_{\text{meat}} + \mathrm{Oa}_{\text{milk}} + \mathrm{Oa}_{\text{egg}} + \mathrm{Oa}_{\text{fish}}\right)/\mathrm{Ia}_{\text{total}}\right] \times 100\,\% \tag{2}$$

where, $\mathrm{Oa}_{\text{meat}}$, $\mathrm{Oa}_{\text{milk}}$, Oa_{egg} and $\mathrm{Oa}_{\text{fish}}$ are the total P outputs in produced meat, milk, egg and fish, respectively, and $\mathrm{Ia}_{\text{total}}$ is the total P input via animal feed. Fish is an output of aquaculture here.P use efficiency in the whole food chain (PUEf) was defined as:

$$\mathrm{PUEf} = \left[\left(\mathrm{Ih}_{\text{plant food}} + \mathrm{Ih}_{\text{animal food}} + \mathrm{Ifp}_{\text{export}} - \mathrm{Ifp}_{\text{import}}\right) \Big/ \left(\mathrm{Ic}_{\text{fertilizer}} + \mathrm{Ia}_{\text{feed import}}\right)\right] \times 100\,\% \tag{3}$$

where, $\mathrm{Ih}_{\text{plant food}}$ is the P in the plant derived food entering households, $\mathrm{Ih}_{\text{animal food}}$ is the P in the animal derived food entering households, $\mathrm{Ifp}_{\text{export}}$ is the total amount of P in exported food, $\mathrm{Ifp}_{\text{import}}$ is the total amount of P in imported food, $\mathrm{Ic}_{\text{fertilizer}}$ is the total P fertilizer use in crop production, and $\mathrm{Ia}_{\text{feed import}}$ is the total amount P in imported animal feed from outside the food production–consumption chain, including P supplementation to feed. The P import in households via detergents and chemicals was not considered in this study.

Socio-economic driving forces

Changes in P use and losses over time were related to driving forces, i.e., economic growth (GDP) and consumption patterns. The following three factors were statistically related to P use and losses: (1) the average GDP per capita, (2) the percentage of animal derived P in human diets, and (3) the percentage of the vegetable and fruit cultivated area (relative to the total cultivated area). Data related to driving forces were derived from NBSC (2009, 2013).

Description of scenarios for 2030

Business as usual (BAU)

Total food requirement in 2030 was based on the expected human population and the average food consumption per capita for urban and rural populations (Table S1). The percentage of animal-derived protein is expected to increase from 34 % in 2005 to 50 % in 2030 (UNDP 2009). As a result, the total demand of crop and animal-derived food will increase by 25 and 80 % between 2005 and 2030, respectively. At the same time, we expect that yields of maize and soybean will increase by 40 % and those of other crops by 10 % due to technological progress (Chen et al. 2014; Ma et al. 2013a). These expected changes were applied to all scenarios as described further below.

Scenario 1: soil and crop management

This scenario builds on the BAU scenario, but assumes balanced P fertilization in crop production, i.e., the P inputs to crops via mineral fertilizers and manures applied matches the demand of the crops, which is a function of crop type, crop yield and soil P level. First, we estimated the amount of P needed to build the soil P status (expressed as P-Olsen) to an agronomic sufficient level, which was set at 20 mg kg^{-1}. Further, we assumed that the Olsen P value increases on average by 3.1 mg kg^{-1} (range 2.1–5.7 mg kg^{-1}, depending on soil type) in the top 20 cm of soil of arable land per 100 kg of P surplus (Li et al. 2011). When the soil P status had reached the level considered sufficient (20 mg kg^{-1}), the total P application rate was set equal to the P withdrawal with harvested crop (edible parts and residues) (Ma et al. 2013a).

Scenario 2: animal production management

This scenario also builds on the BAU scenario, but we assumed that the P content of the animal feed in 2030 can be reduced by on average 20 % compared with

BAU through precision animal feeding (Wang et al. 2011).

Scenario 3: manure management

This scenario also builds on BAU, but we assumed that the percentage of P in manure utilized in cropland increases from an average of 40 % in 2005 to an average of 95 % in 2030. Further, we assumed that the manure P applied to crop land had a similar effectiveness as fertilizer P, and that the manure P applied reduced the requirement of P fertilizer in crop production proportionally (see Table S8).

Scenario 4: diet management

This scenario is based on the recommendations by the Chinese food dietary guidelines (CDG) and the nutrient requirement standards of WTO. As a result, we assume that the fruit, bean, milk and egg consumption will increase, while the meat consumption will decrease compared to 2005. These changes in food consumption patterns were translated in changes in crop and animal production (see Tables S1-S3).

Scenario 5: integration of S1–S4

This scenario is a combination and integration of the scenarios for balanced P fertilization in crop production (S1), precision P feeding in animal production (S2), improved manure management (S3), and adjusted diet management (S4). We assumed that these scenarios can be supplemented simultaneously to the business as usual scenario.

Results

Changes in P input during the period 1950–2010

There was no chemical P fertilizer input before 1960; animal manure, human excreta and crop residues were the main sources of P input to crop land in the 1950s (Fig. 2 and Figure S2). From 1960 to 1980, mineral P fertilizer increased quickly. However, the increase was smaller between 1960 and 1980 than between 1980 and 2010. Concomitantly, the recycling of manure, excreta and residues to crop land dropped rapidly. The contribution of recycled P to the total P input decreased from around 80 % in the 1950s to less than 46 % in 2010 (Figure S2), despite the fact that manure production increased greatly between 1980 and 2010. Changes in P input and output were different for different sectors of the food chain (Figures S3–S5).

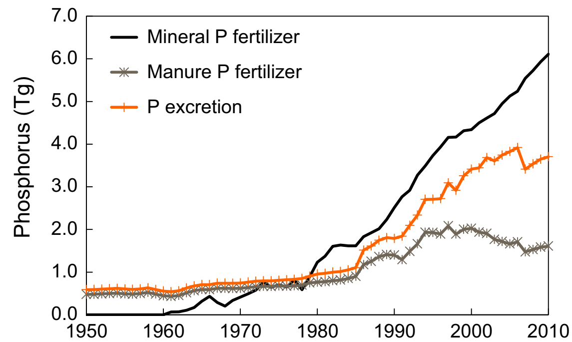


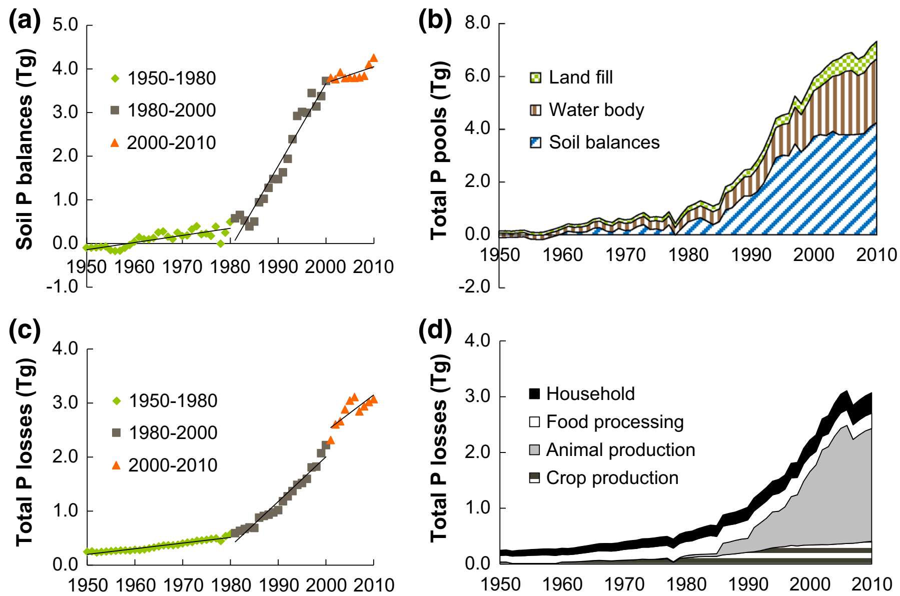
Fig. 2 Estimated inputs of mineral phosphorus (P) fertilizer and manure P fertilizer, and the total P excretion by livestock in China in the period 1950–2010

Changes in P losses and pools during the period 1950–2010

The soil P pool of agriculture land was depleted by about 1.1 Tg P during the period 1950–1959 (Fig. 3a). From 1960, total P inputs surpassed total P withdrawal in harvested products, and the soil P pool rapidly increased. The total net P accumulation amounted to 79 Tg P between 1950 and 2010, around half (49 %) accumulated during the last 10 years (Fig. 3a). The annual P accumulation in farmyards and landfills increased from about 0.1 Tg in 1950 to 0.7 Tg in 2010. We estimated that around 16 Tg P has accumulated in animal confinements and landfill during last 60 years (Fig. 3b). Some of this P could be recovered and applied to crop land, but we have no quantitative estimates for this fraction.

Total P losses to surface water bodies increased from 0.2 Tg in 1950 to 2.4 Tg in 2010. Between 1950 and 2010 around 48 Tg of P has been lost to water bodies, and around half (45 %) of the P losses occurred during the last 10 years. Most of the lost P has accumulated in the sediments of streams, rivers, lakes, and coastal seas. The annual loss of P from crop production increased by a factor of 10, from 0.04 Tg in 1950 to 0.4 Tg in 2010 (Fig. 3d). The estimated P loss from animal production increased from negligible

Fig. 3 Mean changes in **a** soil phosphorus (P) balances, **b** P losses to water bodies and P accumulation in soil and land fill, **c** total P losses, and **d** total P losses by different sectors, during the period 1950–2010. *Negative values* in figures **a**, **b** indicate soil P mining, *positive values* indicates soil P accumulation



small amounts in 1950 to as much as 2.0 Tg in 2010. From the late 1990s, animal production was the biggest contributor to the total P losses.

Changes in PUE during the period 1950–2010

There has been a steady decrease in the P use efficiency in crop production (PUEc) and in the whole food chain (PUEf) between 1950 and 2000 (Fig. 4). Thereafter, PUEc and PUEf remained rather constant, but at a low level. Mean PUEc was 68 % in 1950, peaked in 1956 (89 %), and then dropped to an average of 37 % during the period 2000–2010. Mean PUEf decreased from 35 % in 1950 to 7 % in 2010. In contrast, mean P use efficiency of animal production (PUEa), at animal level, increased from 12 to 17 %

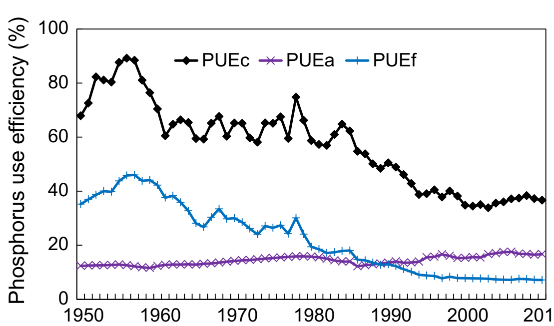


Fig. 4 Changes in P use efficiency in crop production (PUEc), animal production (PUEa) at animal level, and in the food chain (PUEf) in the period 1950–2010

during the period 1950–2010. The increasing PUEa is related to many factors, including a shift from draft animals to monogastric animals and an increased productivity in especially pig and poultry production (Table S3).

Relationships of P fertilizer use, P losses and PUEf with socio-economic drivers

Average GDP increased from 80 to 30,000 Yuan per capita between 1950 and 2010. Both the mineral P fertilizer use and P losses were positive correlated to the GDP, and the PUEf showed a reverse trend (Fig. 5). Changes in mineral P fertilizer use, PUEf and total P losses became decoupled from changes in GDP, when the GDP surpassed a level of about 5000–10,000 Yuan per capita. Relationships between GDP and the percentage of animal-derived food consumption, and between GDP and the percentage of the total cultivated area by cash crops show a very sharp decoupling when the GDP surpassed 10,000 Yuan per capita (Fig. 6).

Changes in mineral P fertilizer use, PUEf, and P losses were also related to the percentage of animal derived P in human diets (Fig. 5d, e, f). The percentage of animal derived P in human diets increased from 8 % in 1950 to 22 % in 2010. As the percentage of animal-derived P in the diet increased, more animal feed and more animals were needed, which boosted mineral fertilizer P use and manure production,

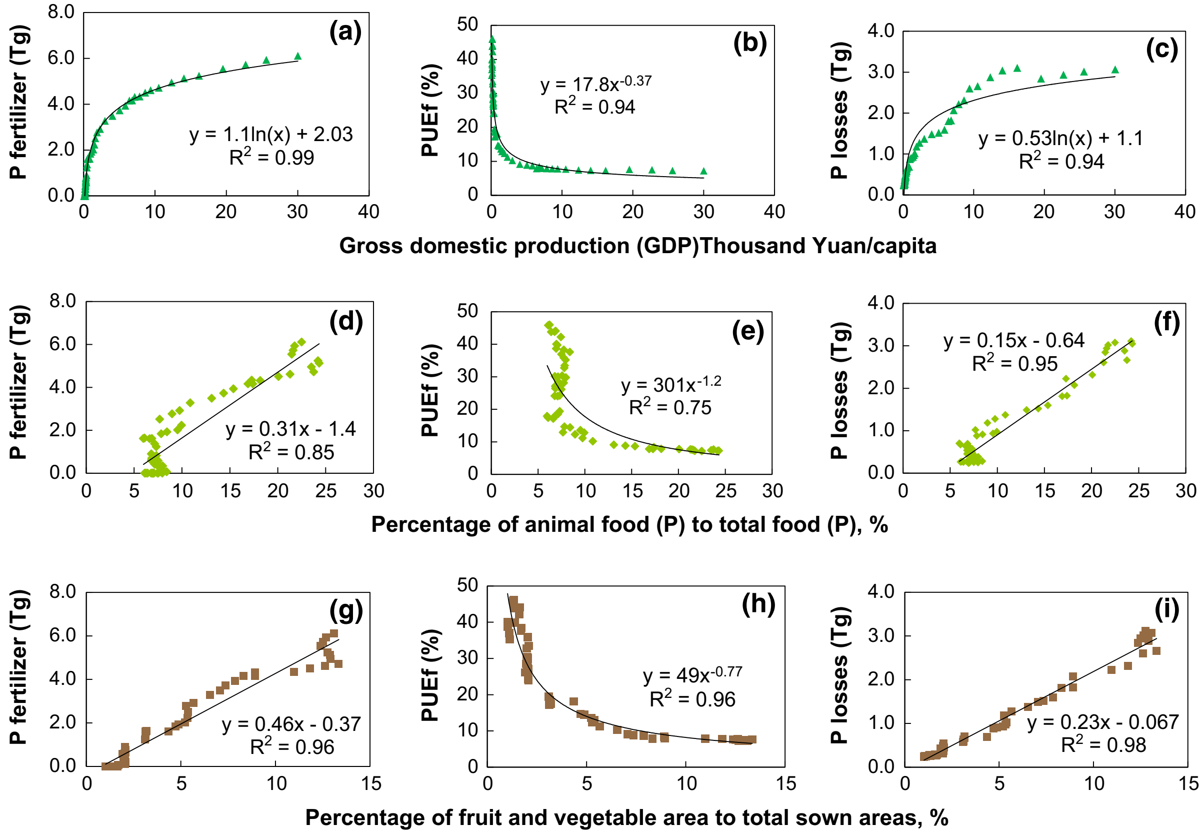


Fig. 5 Relationships between gross domestic production per capita (GDP) and **a** P fertilizer use, **b** phosphorus use efficiency in the food chain (PUEf) and **c** total P losses (*upper panel*). Relationships between the percentage of animal-derived P in food and **d** P fertilizer use, **e** (PUEf) and **f** total P losses (*middle panel*). Relationships between the percentage of fruit and vegetables cultivated areas and **g** P fertilizer use, **h** (PUEf) and **i** total P losses (*bottom panel*)

respectively. Figure 3d clearly shows that animal production has become the main source of P losses, and this trend is fueled by the increasing consumption of animal derived food.

The changes in mineral P fertilizer use, PUEf and P losses were also correlated to the percentage of vegetable and fruit consumption (Fig. 5g, h, i). These relationships follow in part from the very high mineral fertilizer P and manure P application rates in vegetable and fruit production in China (Yan et al. 2013).

Scenario analyses of P use and losses for the year 2030

In the BAU scenario for 2030, mineral P fertilizer use has increased to 6.4 Tg, i.e., 5 % larger than in 2010 (Fig. 7a). Due to the increased consumption of animal products, vegetables and fruits, PUEf is expected to decrease to 6 %, i.e., 18 % less than in 2010. The PUEc is expected to decrease slightly, while PUEa is expected to remain constant between 2010 and 2030 (Fig. 7b). The estimated total P losses increase to 5.1 Tg in 2030, i.e., 66 % larger than in 2010.

Four options and the integration of these options have been identified as pathways to improve the P use efficiency and to decrease P losses in the food chain. Single options are less effective than the integration of all four options. Balanced P fertilization (S1) in crop production decreased mineral fertilizer P input, and increased PUEc (Fig. 7). Precision animal feeding (S2) decreased the P input in animal production and thereby the amounts of P in animal manure and the P losses from animal production. Improved animal manure management (S3) greatly decreased P losses from animal production and at the same time decreased the need for mineral fertilizer P input.

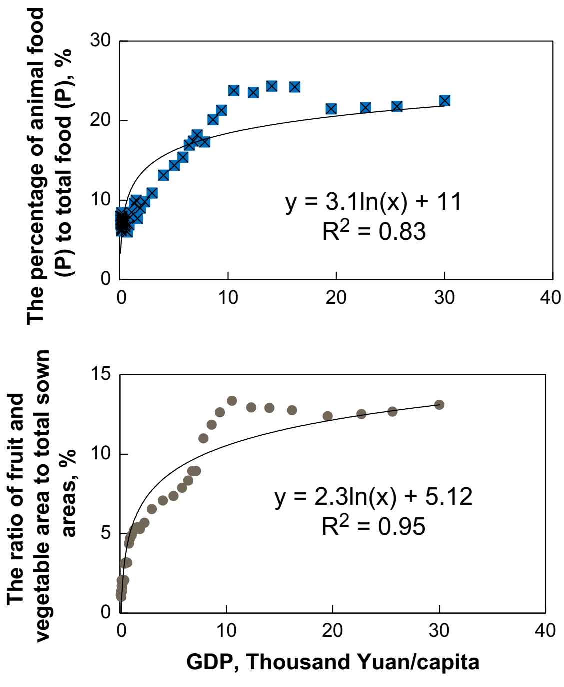


Fig. 6 Relationships between gross domestic production (GDP) and percentage of animal-derived phosphorus (P) in human diets (*upper figure*), and between GDP and percentage of the vegetables and fruit cultivated area (*bottom figure*)

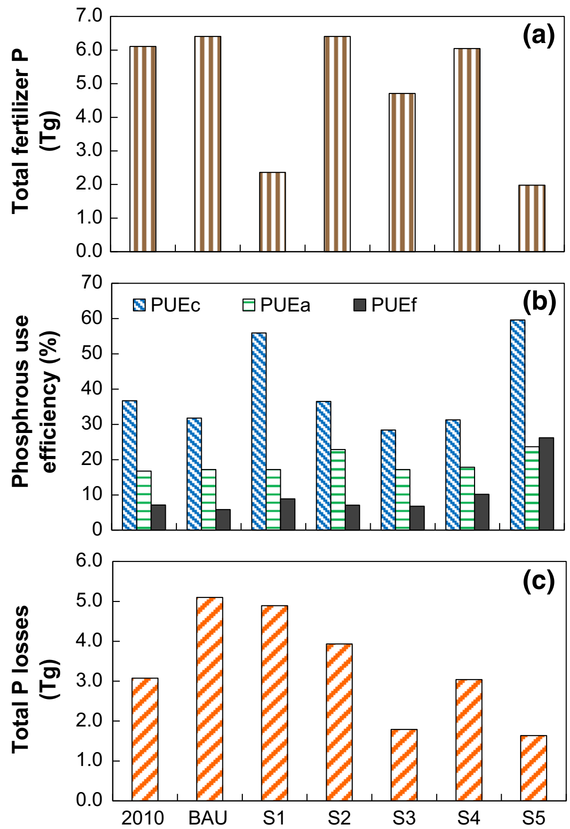


Fig. 7 Forecasts of **a** annual P fertilizer consumption, **b** P use efficiency in crop production (PUEc), animal production (PUEa) and in the food chain (PUEf), and **c** total P losses in 2030, according to six scenarios (*BAU*, *S1*, *S2*, *S3*, *S4*, *S5*). For reference purpose also the values for the year 2010 have been presented (see text). Scenarios: *BAU* business as usual, *S1* balanced P fertilization in crop production, *S2* precision animal P feeding, i.e., lowering the P supplementation of animal feed by 20 %, *S3* improved manure management, i.e., 95 % of manure P is collected and applied to crop land, *S4* compliance to Chinese dietary recommendations, *S5* integration of S1–S4 scenarios

Compliance with the recommendations of a 'healthy diet' (S4) will lead to more vegetable and fruit production and to less animal production; as a result manure P production, mineral fertilizer P use, and P losses will decrease. The integration of the four options reduced mineral P fertilizer requirement by 69 % and total P losses by 68 % (Fig. 7a, c). Also, PUEc increased by 88 %, PUEa by 38 %, and PUEf by 350 %, compared to BAU (Fig. 7b).

Discussion and conclusions

Socio-economic drivers of P fertilizer use

Mineral P fertilizer use was positively related to GDP, till the turn of the century. The use of fertilizer P in China has been facilitated by the booming economy and governmental subsidies to the mineral P fertilizer production and transportation sectors (Zhang et al. 2008; Li et al. 2013). The relationship between GDP and mineral fertilizer P use is confounded by the relationships between GDP and the percentage of animal derived food in the diet (Fig. 6). The changes in the diet to more animal-derived products were clearly positively correlated to mineral P fertilizer use (Fig. 5). Such relationships have been found also for the world (Metson et al. 2012), and for some other countries (Schmid et al. 2008).

Fertilizer P use was also positively related to the areas of vegetable and fruit production (Fig. 5). Farm surveys have indicated that manure and mineral fertilizer application rates are often exceptionally high in vegetable and fruit production (e.g., Yan et al. 2013). Crop land used for vegetable production covered around 12 % of the total crop area in China in 2009, but was responsible for 36 % of the mineral P fertilizer use and 38 % of the manure use (Yan et al. 2013).

The pattern of mineral P fertilizer use in China contrasts with that in Europe and North America. Countries in the European Union (EU) have faced a decrease in mineral fertilizer P use from the 1970s, in response to the increased soil P status, and the increased replacement of mineral P fertilizer by animal manure, which is incentivized by environmental regulations (Withers et al. 2015). Such changes may also happen in China in the future. Our estimates suggest that some 79 Tg P has accumulated in cropland, and as a result soil Olsen-P has increased to about 25 mg/kg by the end of 2010, but with considerable spatial variations (Li et al. 2011; Bai et al. 2013). It is well-known that the need for P input decreases when soil P status increases (e.g., Syers et al. 2008). Recently, the Chinese government initiated several polices aimed to reduce mineral fertilizer input. For example, in 2005 the government started the 'Soil testing and fertilization' project, with the aim to teach farmers to apply fertilizers more rationally (Ma et al. 2013b). In early 2015, a project started with the purpose to achieve a 'zero increasing use of mineral NP fertilizer use by 2020 (MOA 2015).

Main drivers of PUEf

PUEf was negatively related to GDP, and to the consumption of animal-derived food, vegetables and fruits (Fig. 5). PUEf is a resultant of PUEc and PUEa, and the relative proportion of crop and animal derived food in the diet. PUEf has decreased over time, mainly because of the decrease in PUEc and the increase in the percentage of animal-derived food. The increasing animal production sector largely explains the rapid decrease of PUEf, because PUEa was much lower than PUEc, despite the relative increase in PUEa over time (Fig. 4). The shift from mixed crop-livestock production systems to landless animal production systems between 1980 and 2010 (MOA 2013) has also contributed to the decrease of PUEf, as landless animal production systems have limited opportunity for recycling of animal manures in crop production (Ma et al. 2013a).

In the EU and North America, there are strict policy regulations related to the collection, storage and subsequent application of animal manure to crop land during the growing season (Oenema 2004; Sharpley and Beegle 2001). Such regulations contribute to the replacement of mineral fertilizer P by manure P, to low P losses and to a relatively high PUEf (Ott and Rechberger 2012; Suh and Yee 2011; Desmidt et al. 2015).

Main drivers of P losses

Clearly, total P losses were related to GDP and the composition of the human diet (Fig. 5). Such relationships have been found also for other countries (e.g., van Drecht et al. 2009; Schmid et al. 2008). Total annual P losses from the food chain to water bodies have increased to more than 2 Tg in 2010, and most of this P originated from animal production (Fig. 4). Recently, the safe planetary boundary of global P losses to surface waters was set at 6.2 Tg P per year (Carpenter and Bennett 2011; Steffen et al. 2015). This would indicate that losses from the food chain in China in 2010 alone contributed already one-third to this target value. There were several reasons why manure was not applied to arable land but loss to environment in China. Firstly, the high subsidy of P fertilizer production and transportation by the government to ensure the food security, so application of chemical P fertilizer was cheaper compare with applying manure. Secondly, farmers were not well educated, and have little knowledge about the balance P fertilizer application. Thirdly, disappear of local extension services, most of the extension services were done by the fertilizer retailers, who are aiming to profit from over application of chemical fertilizer. Fourthly, development of industrial animal production systems, which has no enough land to applied the manure.

Several studies have indicated that animal manures are the main source of P in surface waters in China (e.g., Ma et al. 2010; Wu et al. 2015). In contrast, animal manures are not considered to be a main source of eutrophication of surface waters in EU and US, despite the importance of animal production in these regions, mainly because the discharge of manures to surface waters is forbidden and enforced in EU and US. Soil erosion and effluents from sewage treatment are seen as the dominant sources of P in surface waters in the EU (Ott and Rechberger 2012). Erosion, P rock mining, and mineral fertilizer P manufacturing accounted for around 70 % of the P in surface waters in the US (Suh and Yee 2011). Summarizing, governmental regulations related to the collection, storage and use of animal manures are a strong driver for minimizing P losses from animal manure in EU and US, but not yet in China. This difference largely explains the differences between these countries in main sources of P losses to surface waters.

Scenarios for improving P use efficiency

Compliance with dietary guidelines will increase PUEf by 75 % compared to the PUEf of the BAU scenario (Fig. 7). This result confirms findings of Ma et al. (2013a), Metson et al. (2012) and Tilman and Clark (2014) that changes in human diets have large impacts on P use and losses. However, complying with dietary guidelines is difficult to achieve in practice; it should be considered as a long-term strategy.

A rather easy applicable option is reducing the P supplementation of animal feed for especially pigs and poultry in large animal confinements. Such a measure could be implemented via a regulation or a tax on P supplements. Reducing the mean P content of the animal feed of pigs and poultry by 20 % will decrease the input of P in the food chain by 9 % (Table S5).

Introducing balanced P fertilization in crop production, especially in cash crop production, would reduce mineral P fertilizer input by 4.0 Tg (63 %), and increase PUEc to 56 %. Implementing this option in practice is again not easy (Ma et al. 2013b); it would require extensive support from extension services, and possibly the withdrawal of subsidies on mineral P fertilizer.

Improving manure management to such extent that essentially all manure P is collected and applied to crop land is also difficult to implement in practice at short notice. This is related the fact that crop and animal production systems have become more and more spatially separated, from the 1980s onwards (Wang et al. 2010; Ma et al. 2012b). Yet, the results of our scenario analysis indicate that this option has huge positive impacts; it will reduce total P losses by 3.3 Tg (65 % reduction) and mineral P fertilizer input by 1.7 Tg (26 % reduction). Increased recycling of manure P is the only sustainable way forward, because dumping or discharging manure in landfill, lagoons or surface waters is environmentally unsustainable (Carpenter and Bennett 2011; Steffen et al. 2015).

Evidently, the combination of all four options has the biggest effect; it would reduce mineral P fertilizer input by up to 4.4 Tg (69 % reduction), and increase PUEf in the whole food chain from 6 % in the BAU scenario to 26 % in the combined options scenario. Single options were less effective in reducing both mineral P fertilizer use and P losses than the integration of the four options. This finding is supported by other studies, which concluded that no single option was effective in reducing P losses and mineral fertilizer P use simultaneously (Cordell et al. 2011; Withers et al. 2015).

In conclusion, an integrated mitigation options were identified to improve P management in the food production–consumption chain. The increase in animal production and vegetable production from the 1980s onwards has greatly contributed to the increased P accumulation in soils, the increased P losses to land fill and surface waters, and to a decrease in the P use efficiency (PUEf), as they were linear related. Our scenario analysis for the year 2030 indicate that mineral P fertilizer use and total P losses will greatly decrease, and PUEf greatly increase through a combination of precision animal feeding, improved manure management, balanced P fertilization and compliance with dietary guidelines. However, the implementation of all four options in practice simultaneously requires huge efforts from both scientists, policy makers, practitioners, farmers and consumers.

The limitations of the study were we rely on statistical data which have some uncertainties themselves. For example, the production data could not be compared to the consumption data according to the statistical data. The P fertilizer application rates between different crops were based on incidental farm surveys carried out during the last 20 years, which also have some uncertainties. Parameters related to food consumption and production between 1950 and 1980 are depending on assumptions. Evidently, more field work needs to be done to be able to improve the accuracy of the P budgets and especially to quantify the fate of the large amounts of P not recovered in the food products.

Acknowledgments This work was financially supported by the Chinese National Basic Research Program (2015CB150405) and the Hundred Talent Program of the Chinese Academy of

Sciences. The authors would like to extend special thanks to Thomas Nesme and two anonymous reviewers of their constructive comments and suggestions in improving the quality of this paper.

References

Bai Z, Li H, Yang X, Zhou B, Shi X, Wang B, Li D, Shen J, Chen Q, Qin W, Oenema O, Zhang F (2013) The critical soil P levels for crop yield, soil fertility and environmental safety in different soil types. Plant Soil 372(1–2):27–37

Carpenter SR (2008) Phosphorus control is critical to mitigating eutrophication. PNAS 105:11039–11040

Carpenter SR, Bennett EM (2011) Reconsideration of the planetary boundary for phosphorus. Environ Res Lett 6(1):014009

Carstensen J, Andersen JH, Gustafsson BG, Conley DJ (2014) Deoxygenation of the Baltic Sea during the last century. PNAS 111(15):5628–5633

Chen X, Cui Z, Fan M, Vitousek P, Zhao M, Ma W, Wang Z, Zhang W, Yan X, Yang J, Deng X, Gao Q, Zhang Q, Guo S, Ren J, Li S, Ye Y, Wang Z, Huang J, Tang Q, Sun Y, Peng X, Zhang J, He M, Zhu Y, Xue J, Wang G, Wu L, An N, Wu L, Ma L, Zhang W, Zhang F (2014) Producing more grain with lower environmental costs. Nature 514(7523):486–489

Conley DJ, Paerl HW, Howarth RW, Boesch DF, Seitzinger SP, Havens KE, Lancelot C, Likens GE (2009) Controlling eutrophication: nitrogen and phosphorus. Science 323(5917):1014–1015

Cordell D, Drangert JO, White S (2009) The story of phosphorus: global food security and food for thought. Glob Environ Change 19(2):292–305

Cordell D, Rosemarin A, Schröder JJ, Smit AL (2011) Towards global phosphorus security: a systems framework for phosphorus recovery and reuse options. Chemosphere 84(6):747–758

Desmidt E, Ghyselbrecht K, Zhang Y, Pinoy L, Van der Bruggen B, Verstraete W, Rabaey K, Meesschaert B (2015) Global phosphorus scarcity and full-scale P-recovery techniques: a review. Crit Rev Environ Sci Technol 45(4):336–384

FAO (2014) FAOSTAT. http://faostat.fao.org/site/291/default.aspx. Accessed 2014

Godfray HCJ, Beddington JR, Crute IR et al (2010) Food security: the challenge of feeding 9 billion people. Science 327(5967):812–818

Li H, Huang G, Meng Q, Ma L, Yuan L, Wang F, Zhang W, Cui Z, Shen J, Chen X, Jiang R, Zhang F (2011) Integrated soil and plant phosphorus management for crop and environment in China. A review. Plant Soil 349(1–2):157–167

Li Y, Zhang W, Ma L, Huang G, Oenema O, Zhang F, Dou Z (2013) An analysis of China's fertilizer policies: impacts on the industry, food security, and the environment. J Environ Qual 42(4):972–981

Ma L, Ma WQ, Velthof GL, Wang FH, Qin W, Zhang FS, Oenema O (2010) Modeling nutrient flows in the food chain of China. J Environ Qual 39(4):1279–1289

Ma D, Hu S, Chen D, Li Y (2012a) Substance flow analysis as a tool for the elucidation of anthropogenic phosphorus metabolism in China. J Clean Prod 29:188–198

Ma L, Velthof GL, Wang FH, Qin W, Zhang WF, Liu Z, Zhang Y, Wei J, Lesschen JP, Ma WQ, Oenema O, Zhang F (2012b) Nitrogen and phosphorus use efficiencies and losses in the food chain in China at regional scales in 1980 and 2005. Sci Total Environ 434:51–61

Ma L, Wang F, Zhang W, Ma W, Velthof G, Qin W, Oenema O, Zhang F (2013a) Environmental assessment of management options for nutrient flows in the food chain in China. Environ Sci Technol 47(13):7260–7268

Ma L, Zhang WF, Ma WQ, Velthof GL, Oenema O, Zhang FS (2013b) An analysis of developments and challenges in nutrient management in China. J Environ Qual 42(4):962–971

MacDonald GK, Bennett EM, Potter PA, Ramankutty N (2011) Agronomic phosphorus imbalances across the world's croplands. PNAS 108(7):3086–3091

Metson GS, Bennett EM, Elser JJ (2012) The role of diet in phosphorus demand. Environ Res Lett 7(4):044043

MOA (Minister of Agriculture) (2009) New China's agricultural statistics for 60 years. China Statistics Press, Beijing

MOA (Minister of Agriculture) (2013) China husbandry yearbook. China Statistics Press, Beijing

MOA (Minister of Agriculture) (2015) http://www.moa.gov.cn/zwllm/tzgg/tz/201503/t20150318_4444765.htm. Accessed 2015.03. (In Chinese)

NBSC (National Bureau of Statistics of China) (2009) China Compendium of Statistic 1949–2008. China Statistics Press, Beijing

NBSC (National Bureau of Statistics of China) (2013) China statistic yearbook. China Statistics Press, Beijing

Oenema O (2004) Governmental policies and measures regulating nitrogen and phosphorus from animal manure in European agriculture. J Anim Sci 82(13_suppl):E196–E206

Ott C, Rechberger H (2012) The European phosphorus balance. Resour Conserv Recycl 60:159–172

Richardson AE (2009) Regulating the phosphorus nutrition of plants: molecular biology meeting agronomic needs. Plant Soil 322:17–24

Schmid NTS, Bader HP, Scheidegger R, Lohm U (2008) The flow of phosphorus in food production and consumption—Linköping, Sweden, 1870–2000. Sci Total Environ 396(2):111–120

Sharpley AN, Beegle D (2001) Managing phosphorus for agriculture and the environment. College of Agricultural Sciences, The Pennsylvania State University, University Park, PA

Smil V (2000) Phosphorus in the environment: natural flows and human interferences. Annu Rev Energy Environ 25:53–88

Steffen W, Richardson K, Rockström J, Cornell SE, Fetzer I, Bennett EM, Biggs R, Carpenter SR, de Vires W, de Wit CA, Folke C, Gerten D, Heinke J, Mace GM, Persson LM, Ramanathan V, Reyers B, Sörlin S (2015) Planetary boundaries: guiding human development on a changing planet. Science 347(6223):1259855

Suh S, Yee S (2011) Phosphorus use-efficiency of agriculture and food system in the US. Chemosphere 84(6):806–813

Syers JK, Johnston AE, Curtin D (2008) Efficiency of soil and fertiliser phosphorus use: reconciling changing concepts of soil phosphorus behaviour with agronomic information. Food and Agriculture Organization of the United Nations Report, Rome

Tilman D, Clark M (2014) Global diets link environmental sustainability and human health. Nature 515(7528): 518–522

UN (United Nations) (2010) World population prospects: the 2008 revision population database

UNDP (2009) China and a sustainable future: towards a low carbon economy and society, china human development report 2009/10. China Translation & Publishing Corporation: Beijing, 2010

van Drecht G, Bouwman AF, Harrison J, Knoop JM (2009) Global nitrogen and phosphate in urban wastewater for the period 1970 to 2050. Global Biogeochem Cycles 23(4)

Wang F, Dou Z, Ma L, Ma W, Sims JT, Zhang F (2010) Nitrogen mass flow in China's animal production system and environmental implications. J Environ Qual 39(5): 1537–1544

Wang F, Sims JT, Ma L, Ma W, Dou Z, Zhang F (2011) The phosphorus footprint of China's food chain: implications for food security, natural resource management, and environmental quality. J Environ Qual 40(4):1081–1089

Withers PJA, Van Dijk K, Neset TS (2015) A 5R stewardship to tackle global phosphorus inefficiency: the case of Europe. Ambio 44(2):193–206

Wu H, Yuan Z, Gao L, Zhang L, Zhang Y (2015) Life-cycle phosphorus management of the crop production–consumption system in China, 1980–2012. Sci Total Environ 502:706–721

Yan Z, Liu P, Li Y, Ma L, Alva A, Dou Z, Chen Q, Zhang F (2013) Phosphorus in China's intensive vegetable production systems: over fertilization, soil enrichment, and environmental implications. J Environ Qual 42(4): 982–989

Zhang W, Ma W, Ji Y, Fan M, Oenema O, Zhang F (2008) Efficiency, economics, and environmental implications of phosphorus resource use and the fertilizer industry in China. Nutr Cycl Agroecosyst 80(2):131–144