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PERSPECTIVE

Identifying potential strategies in the key sectors of China's food chain to implement sustainable phosphorus management: a review

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Abstract High extraction of phosphate reserves and low phosphorus utilization efficiency in the food chain in China result in large P losses and serious environmental pollution. The P fertilizer industry, soil P surplus, livestock manure P and wastewater P recycling have been identified as the priority sectors based on summarizing several systemic and in-depth reviews of P flows analysis. Mineral P fertilizer production has reached 7.4 Mt P in 2012, which is more than seven times the value in 1980. The large P surpluses in arable land resulted in soil P accumulation of up to 64 Mt during the period 1951–2010. Livestock numbers have increased dramatically (more than ten times) during the period 1949–2012 in China, especially pigs and poultry, and so has the quantity of manure that they produce. The average loading of manure P on arable land in China has increased significantly from 9.5 kg P ha^{-1} in 1980 to 20.4 kg P ha^{-1} in 2010. Up to 0.49 Mt of wastewater P discharged without treatment also exerted great pressure on the environment in 2012. Based on an understanding of P interactions in these key sectors, an integrated set of policy options and technical measures is proposed. Taking P flows in China in 2010 as an example, if all of the strategies recommended in this study are adopted in P management, about 4.3, 2.5, 1.6 and 0.3 Mt of P resources, respectively, will be saved in the P fertilizer industry, arable land production, livestock manure and wastewater sectors.

Keywords Phosphate reserves · Phosphorus surplus · Manure · Wastewater · Management strategies

Introduction

Phosphorus (P) is a finite natural resource and is a major limiting nutrient in agriculture in many parts of the world (Sattari et al. 2012; MacDonald et al. 2011; Syers et al. 2011). The growing consumption of P fertilizers from mining of nonrenewable phosphate rock has contributed to the production of food for the nearly seven billion people on earth (Tilman et al. 2002; Lott et al. 2011). Because P is a major essential nutrient for plants with no substitute in food production (Epstein 1972), the rapidly depleting world P reserves are increasingly considered to be a new global sustainability challenge of the twenty-first century (Cordell et al. 2009; Cordell and Neset 2014).

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However, large P losses and environmental pollution due to high extraction of phosphate rock and low P utilization efficiency (PUE—see "Materials and methods" section for a precise definition) have recently become a prevalent problem in many countries (Vaccari 2009; Sun et al. 2012; Zhang et al. 2004). Hence, a more efficient utilization of P reserves is needed to delay the impact of dwindling amounts of extractable P and increasing costs of P fertilizer production on food security.

Currently, material flow analysis (MFA), a complete, systematic and in-depth approach, is widely used to understand the nature and magnitude of P flows through different systems at different geographical scales (Liu et al. 2008; Villalba et al. 2008; Cordell et al. 2009, 2013; Ott and Rechberger 2012; Antikainen et al. 2005; Belevi 2002; Chen et al. 2008; Cooper and Carliell-Marquet 2013; Cordell and White 2013, 2014; Jeong et al. 2009; Matsubae-Yokoyama et al. 2009; Senthilkumar et al. 2012; Suh and Yee 2011; Wu et al. 2014; Qiao et al. 2011; Schmid Neset et al. 2008; Yuan et al. 2011a, b; Bi et al. 2013; Li et al. 2010). A holistic MFA of all sectors from rock mining to food consumption at country scale is described in Fig. 1 according to a comprehensive understanding of anthropogenic flows of P.

The MFA approach has often successfully traced P pathways and identified the key sectors causing environmental pollution based on the P flows structure shown in Fig. 1 (Cordell et al. 2013; Chen et al. 2010). Most of the research results have demonstrated that low PUE in the food chain, especially in agricultural and livestock production systems, results in a massive P surplus in some agricultural soils and in manure P losses. Both of these act as drivers of eutrophication in freshwater and coastal systems (Bennett et al. 2001; MacDonald et al. 2011; Rabalais et al. 2010). Suh and Yee (2011) found that P run-off from livestock waste and crop land are the two largest contributors to total life-cycle P loss in the US food production-consumption chain. Based on our rough recalculation of data in the literature, PUE in the P flow chain analyzed by the MFA approach at different geographical scales was about 13 % in Europe as a whole (Ott and Rechberger 2012), 10 % in France (Senthilkumar et al. 2012), 11 % in Finland (Antikainen et al. 2005), 19 % in Japan (Matsubae-Yokoyama et al. 2009), 12 % in the UK (Cooper and Carliell-Marquet 2013), 21 % in South Korea (Jeong et al. 2009), and 7 % in China (Ma et al. 2010). Common trends in these studies were relatively low PUE values of livestock production and crop cultivation and a lower ratio of P recycling from wastewater, resulting in very large losses of P to the environment. These sectors are considered to be the key sectors where some additional management measures will lead to a significant improvement in the PUE of the food production-consumption chain.

China is a large agricultural country feeding 22 % of the global population (1.3 billion) with <7 % of the global arable land (arable + permanent crops) (130 million ha) (Li et al. 2011) and is playing a critical role in global P production and consumption trends (Sattari et al. 2014). For national food security reasons, the mineral fertilizer-intensive agriculture has developed enormously in China. Mineral P fertilizers in China are mined from native P rock phosphate reserves, and this mining has increased greatly during the last two decades. However, about 70 % of P rock phosphate resources are classified as low grade and the low PUE of P rock mining and processing in the P fertilizer industry sector have aggravated the depletion of native P rock reserves and the associated environmental pollution (Zhang et al. 2008). Also, the inappropriate and spatial imbalance of P fertilizer application has led to massive soil P surplus in arable land and a much lower PUE (about 7 %) in the P flow chain compared with other counties as described above, resulting in an increasing risk of P loss from agricultural land to ground water and surface waters (Liu et al. 2007; Sheng et al. 2003). Concomitantly, huge amounts of nutrient losses from animal housing facilities and a shortage of sewage treatment facilities has resulted in insufficient recycling of P in manures and wastewaters with the result of immense pressure on the environment.

As one of the most important countries in the utilization of global P resources, an improvement in the PUE in the food chain in China will have an important impact on the worldwide use efficiency of P resources. Instead of providing another comprehensive analysis of P flows in China's food chain, we think that a focus on the key sectors with respect to low PUE and environmental pollution could provide more insight in alternative management and policy options. Therefore, in the present study we analyze the key sectors, including the P fertilizer industry, soil P surplus, manures and wastewater P in China to identify the potential options for improved P resource

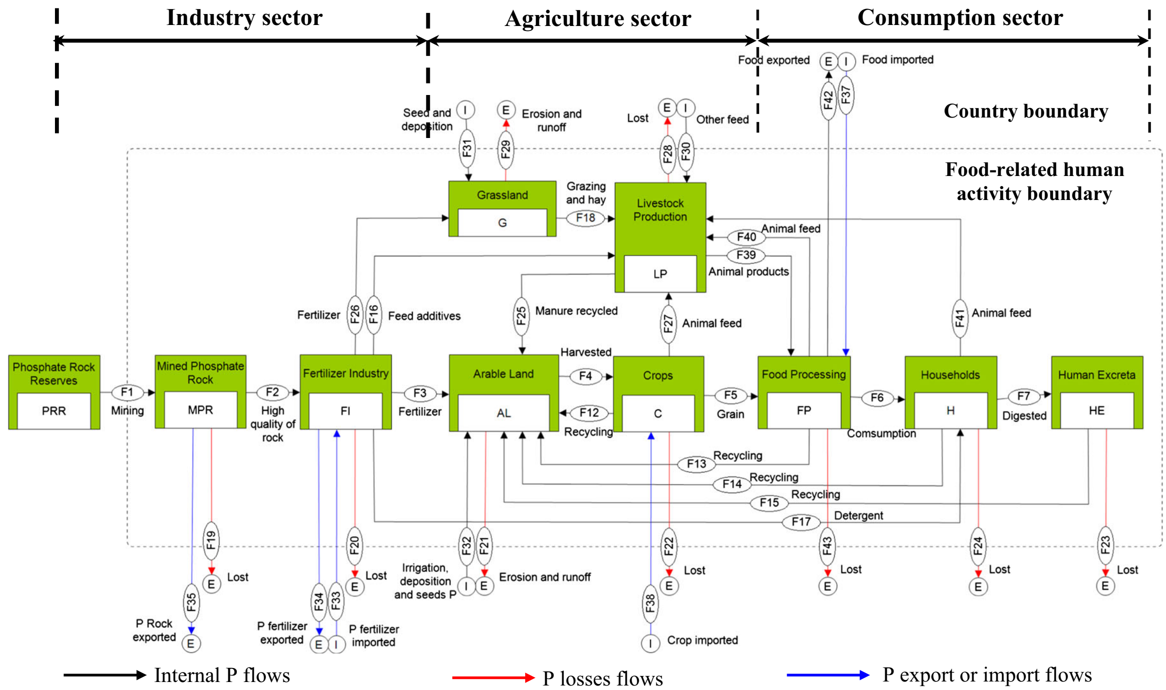

Fig. 1 Key sectors of P flows through the food chain and the intersectoral interactions in terms of P flows indicate P usage, stocks, losses and recycling at each sector of the process. The *green boxes* in the diagram represent the different key sectors which are inter-related through P flows. The industry sector describes mainly the upstream processes of P mining and fertilizer production. The agricultural sector describes the P processes in arable land, grassland and livestock production systems. The consumption sector describes mainly the food processing and household consumption. (Color figure online)

management. Through an in-depth understanding of P flows, we recommend integrated systemic strategies for these key sectors which can be useful for production managers, politicians and policy makers. In summary, the objectives of the study were to quantify the P situation in the key sectors in the Chinese food chain, to identify the potential strategies in these key sectors for sustainable management of P resources, and to evaluate the potential savings of P resources with adoption of the recommended technologies in the key sectors.

Materials and methods

We investigated the developments in P fertilizer industry, manure P, soil P surplus and wastewater P for China. For the P fertilizer industry, the phosphate reserves, the production, consumption, imports and exports of P fertilizers and the structure of mineral P fertilizer products were determined for the period 1980–2013. For manure P, the changes (1949–2012) in the animal stocks and manure P loadings in arable land were investigated. For soil P surplus, the changes (1951–2010) in soil P budget and PUE in arable land were calculated. Finally, for wastewater P, the amount of wastewater P discharged to the environment in 2012 was estimated.

Data sources

Data on phosphate rock reserves were obtained from USGS Phosphate Rock Mineral Commodity Summaries 1995–2012 (http://minerals.usgs.gov/minerals/pubs/commodity/phosphate_rock) and data on P fertilizer production and consumption in China during the period 1980–2012 were taken from the governmental statistical data (http://www.china-npk.org/cn/index.html). Data on P fertilizer imports and exports were derived from China Chemical Information Center (http://www.china-fertinfo.com.cn/hyxx.aspx).

Information on crop yields, arable land areas, wastewater P discharged and number of animals during the period 1950–2012 was derived from authoritative statistical sources (MOA 1949–2012; NBSC 1949–2013; ECCEY 2012). The concentrations of P in harvested crops and crop residues, the crop residue factors (RF), and the ratios of crop residue recycling to P excretion values per animal category were derived from literature (Wang et al. 2006, 2012; Ma et al. 2010; Gao et al. 2009).

Calculation methods

The total amount of P in harvested crops (comprising 16 types of crops, vegetables and fruits) was calculated using Eq. (1);

$$\sum_{1}^{16} P\,in\,harvested\,crops = \sum_{1}^{16}\left[Crop\,yields\,(i) \times C_{yields}(i)\right], \tag{1}$$

where '$C_{yields}(i)$' is the concentration of P in crop yields.

The total amount of P in crop residues recycled to arable land was calculated using Eq. (2);

$$\sum_{1}^{16} P\,in\,recycled\,residues = \sum_{1}^{16}\left[Crop\,yields(i) \times RF(i) \times C_{residues}(i) \times R(i)\right], \tag{2}$$

where '$RF(i)$' is the crop residues factor = weight of crop residues/weight of crop yields); '$C_{residues}(i)$' is the concentration of P in crop residues; and '$R(i)$' is the recycling ratio of crop residues.

Total P inputs in crop production were estimated from the amounts of P in mineral P fertilizers, crop residues recycled to arable land, and manures and the fertilizers (both livestock manures and human excreta), Eq. (3);

$$TP_{inputs} = TP_{mineral\,fertilizer} + TP_{manure} + \sum_{1}^{16} P\,in\,residues\,recycled, \tag{3}$$

Total P outputs in crop production were estimated from the amounts of P in harvested crops and crop residues taken away from arable land for other uses, Eq. (4);

$$TP_{outputs} = \sum_{1}^{16} P\,in\,harvested\,crops + \sum_{1}^{16} P\,in\,residues\,taken\,away, \tag{4}$$

where '$\sum_{1}^{16} P\,in\,residues\,taken\,away$' is calculated based on Eq. (2).

The P utilization efficiency in crop production (PUEc) and soil P surplus in arable land were calculated as in Eqs. (5) and (6), respectively.

$$PUEc = TP_{outputs}/\,TP_{inputs} \times 100\,\%, \tag{5}$$

$$Soil\,P\,surplus = TP_{inputs} - TP_{outputs}, \tag{6}$$

The loading of animal manure P in different Chinese provinces was calculated as in Eq. (7);

$$P_{load} = TP_{animal\,manure}/S_{arable\,land}. \tag{7}$$

where 'P_{load}' is the animal manure P loading in arable land, '$TP_{animal\ manure}$' is the total amount of animal manure P in a specific province, and '$S_{arable\ land}$' is the total area of arable land in the same province.

P fertilizer industry

Until the end of the 1950s arable soil fertility replenishment in China was largely based on the recycling of animal manures and human feces, as well as household wastes and cereal-straw ash. Since then the mineral P fertilizer industry has expanded. The production and consumption of P fertilizers in China have increased greatly, especially from the 1980s (Fig. 2), driven mainly by massive government subsidies to the fertilizer industry [the total subsidy for fertilizer industry was \$18.8 billion in 2010, which translates in \$283 per ton nutrient (N + P + K)] (Li et al. 2013). The production of mineral P fertilizers has increased from 1.0 Mt P in 1980 to 7.4 Mt P in 2012, compared with a consumption increase from 1.0 Mt to 5.2 Mt P at a rate of about 5 % increase per year. Due to restrictions in the industrial production capacity of mineral P fertilizers during 1980–2006, the production of mineral P fertilizers in China could not satisfy the increasing fertilizer consumption demands at home. The gap of millions of tonnes of fertilizers was filled mainly by imports through international trade. For example, the maximum gap of mineral P fertilizers was 3.0 Mt P_2O_5 (1.3 Mt P) in 1995 and about 1.2 Mt P was imported from the US and Russia to fill the gap (Fig. 2).

After entering the twenty-first century the fertilizer enterprises continued to introduce advanced mining equipment and modern production management technologies from abroad, resulting in an increase in P fertilizer production at an average rate of 10.7 % from 2001 to 2007. Consequently, in 2006 China had already become a net exporting country and since then export volumes have increased significantly (Fig. 2). China has now become the largest global consumer and producer of P fertilizers.

With the development of seismic exploration technology the known reserves of phosphate rock deposits have increased from 92 Mt P in 1995 to 1615 Mt P in 2012 (Fig. 2). China has now the second largest phosphate rock deposits in the world, accounting for an estimated 5.5 % of world deposits (USGS 2014). However, about 80 % of the deposits in China are of sedimentary origin, low in quality, and only 7 % has been classified as high-grade P rock resources (30 % P_2O_5) (Zhang et al. 2005, 2008). Figure 2 shows a dramatic increase of phosphate rock reserves in China since 2002. Prior to 2003, due to few available official data, China was thought to be a relatively small phosphate rock reserve holder. However, the Chinese government released its first official phosphate rock deposit data in 2003. Since that time, their reserve estimates have been revised upward by the USGS (Van Kauwenbergh et al. 2013).

About 87.3 % of phosphate rock in China is used for agriculture, 78.3 % as P fertilizer, and 9.0 % as a direct supplement in animal feeds. The remaining 12.7 % is used for yellow P which is normally used in detergents, fire retardants, and food additives (Junfa 2009). Usually, the high-grade phosphate rock (P_2O_5 content >30 % after processing) is used to produce high-quality P fertilizers such as ammonium phosphate. In contrast, low-grade phosphate rock with an average P_2O_5 content of 16 % is used to produce low-grade P fertilizers such as single superphosphate. The production of high-grade P fertilizers has increased dramatically since 1995 as illustrated in Fig. 3, and this has promoted a steep increase in phosphate rock mining and resulted in large amounts of low-grade phosphate rock discharge (Zhang et al. 2008). In 1980 single superphosphate accounted for about 72.4 % of P fertilizer products and the remaining 27.6 %

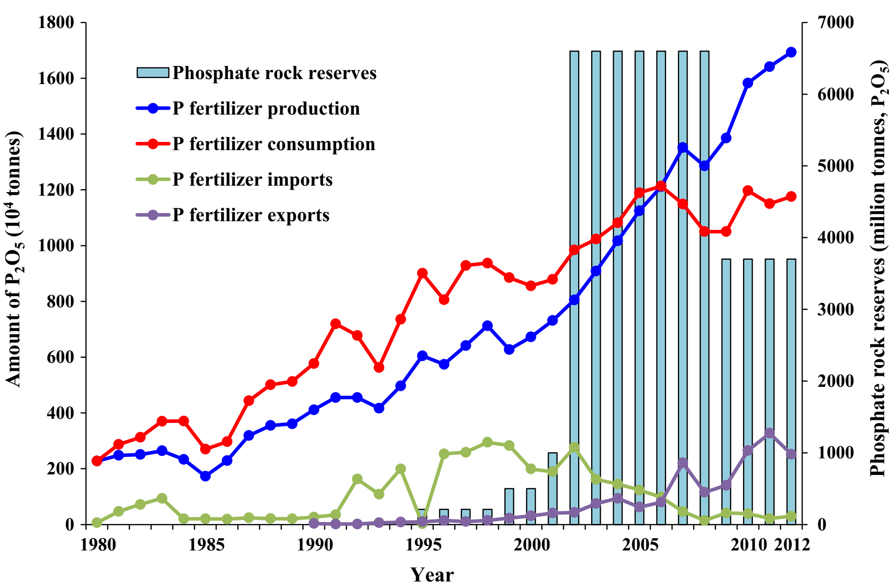

Fig. 2 Changes in phosphate reserves and P fertilizer production, consumption, imports and exports in China from 1980 to 2012

comprised calcium magnesium phosphate products. The quality of Chinese P fertilizer products has improved greatly owing to the great progress achieved by the P fertilizer industry in recent decades. In 2012 the national total P fertilizer production had multiplied by more than seven times the 1980 figure and reached more than 7.4 Mt P, 83.5 % of which consisted of high-grade fertilizers including ammonium phosphate and compound fertilizers (Fig. 3).

The massive increase in phosphate rock mining and P fertilizer production has resulted in severe environmental pollution because of the lack of effective legal controls. It was reported that the PUE of the Chinese P fertilizer industry had decreased from 71 % before 1995 to 39 % in 2003 with 56 % of the residues discarded at the mining sites and 5 % lost as wastes during manufacturing (Zhang et al. 2008). Currently, phosphogypsum (PG) is a by-product generated in the phosphoric acid process for manufacturing fertilizers and is also a large contributor to the environmental problems and land deterioration. The quantity of PG generated is enormous: for each tonne of phosphoric acid made about 4–5 tonnes of PG are created (Smadi et al. 1999; Canut et al. 2008). Until 2011 the total cumulative amount of PG generated in the country was estimated at 250 Mt. The annual production of this material was estimated to be about 70 Mt at the end of the 12th Five-Year Plan (Wang 2013). Because this by-product is composed mainly of high levels of gypsum, phosphates, fluorides, sulphates, heavy metals, and other trace elements, it cannot be reused and is largely disposed without any treatment, by dumping in large stockpiles occupying large areas of land and causing severe environmental damage (Tayibi et al. 2009). In addition, a massive amount of low-grade phosphate rock is also discharged in large stockpiles due to the high costs of remediation processes. It was estimated that these large stocks occupied and deteriorated about 475 km^2 land, and the consumption of ground water was 1.8 billion m^3 per year (Zhang et al. 2008).

Currently, the average PUE of phosphate rock in the Chinese P fertilizer industry is about 61 %, much lower than the 98 % in the USA (Zhang et al. 2008). On the one hand China is suffering from the depletion of its high-quality phosphate rock resource and on the other hand there is severe environmental damage caused by the low utilization efficiency of phosphate

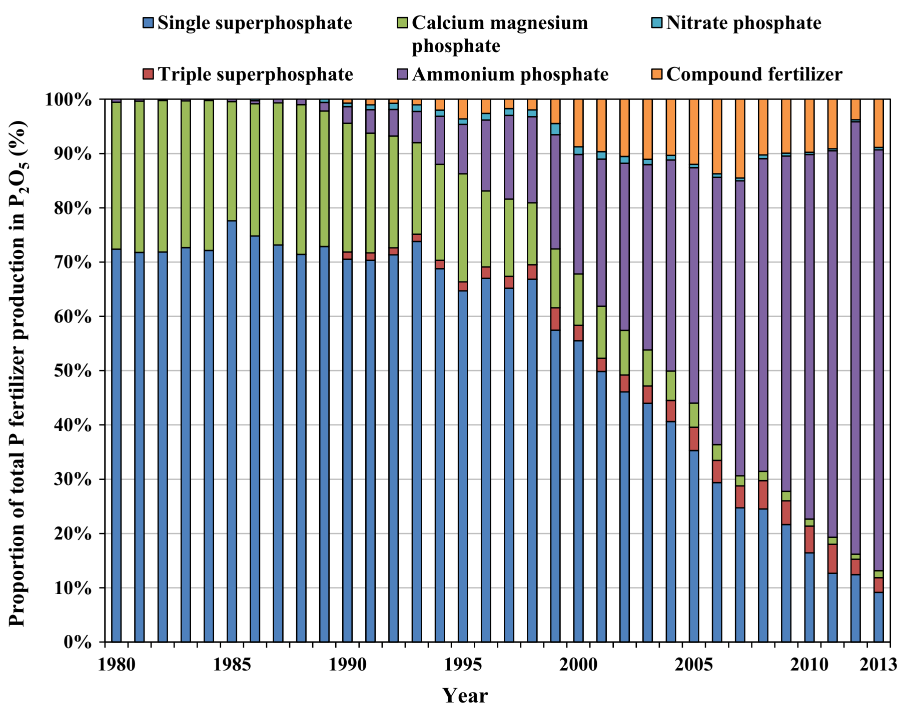

Fig. 3 Changes in the structure of mineral P fertilizer products in China from 1980 to 2013

rock deposits and the irrational management of byproducts. Therefore, increasing the PUE of phosphate rock by new technologies, a better re-utilization of the by-product and developing new and more efficient P fertilizer products by the P fertilizer industry are the potential options for a more sustainable use of this essential resource.

Manure P

Rapid economic growth in China and increasing incomes and changes in dietary structure from cereal-based to more meat-based diets have stimulated the intensive production of livestock in industrial and specialized enterprises in peri-urban areas. Animal stocks in China have been growing substantially since the early 1950s (Fig. 4). For example, the number of pig slaughtered has increased by more than tenfold from 66 million heads in 1952 to 698 million heads in 2012, and the number of other livestock such as poultry and sheep also show similar growth trends (Fig. 4). There are several periods without any poultry stocks due to no available data. The dramatic increase in livestock numbers has led to the generation of large amounts of livestock manures. In this study, the percentage of manure P excretion per animal category in captivity that was recycled to arable land was determined according to data in literature (Ma et al. 2010), while the remainder of manure P was considered to be discarded or lost to the environment. In contrast, manure P excreted by grazing animals was considered to be recycled in grasslands completely. The amount of animal manure P recycled to arable land was considered in the construction of the cropping budgets. From 1952 to 2012, the total amount of manure P excreted by pigs, large animals and sheep has increased by 1066, 158 and 461 %, respectively. The amounts of manure P excreted by poultry and rabbits have also increased by 765 and 676 %, respectively, in 2012 compared with 1986. In 2012 the total amounts of manure P excreted by pigs, large animals, sheep and poultry were estimated at about 1179, 574, 285 and 242 ($\times 10^3$ t), respectively (data not shown). More than 48 % of total P in pig manures, 87 % of total P in dairy manures and 69 % of total P in poultry manures was in water-soluble forms which are plant-available and readily lost to the environment (Li et al. 2014).

At the national level the average loading of livestock manure P in arable land (amount of animal manure P excreted divided by the total cropland area) has increased gradually, i.e. 9.5 kg P ha^{-1} in 1980, 14.8 kg P ha^{-1} in 1990, 17.8 kg P ha^{-1} in 2000 and 20.4 kg P ha^{-1} in 2010 (Fig. 5). There is a wide range of livestock manure P loadings in arable land in different provinces. Across all decades the highest manure P loading occurs in Xinjiang and Qinghai

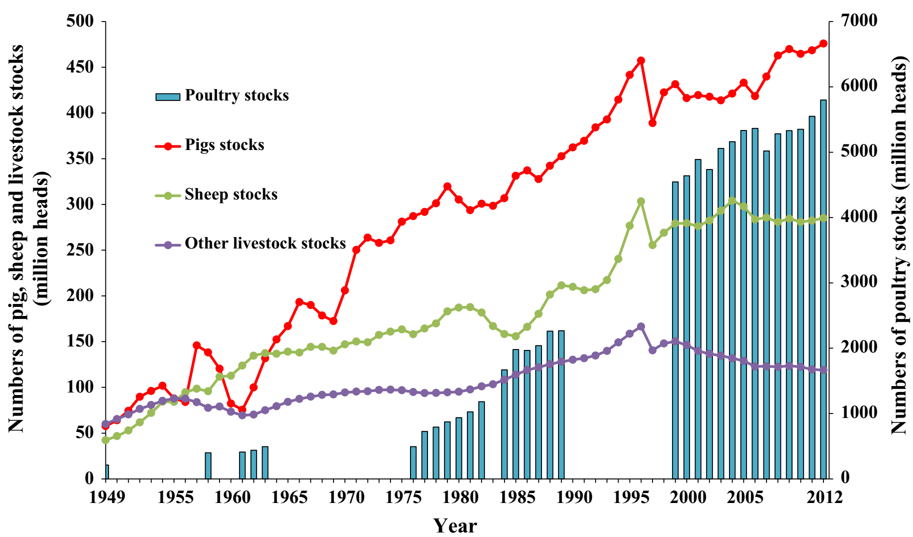

Fig. 4 Past trends in animal numbers in China from 1949 to 2012. Other livestock includes cattle, donkeys, horses, camels and mules

provinces at more than 40 kg P ha^{-1}. One possible explanation is that there is less arable land in these two provinces but with large numbers of animals. In the 1980s and 1990s the leading region of Chinese livestock breeding was in the southwest, mainly in Sichuan, Yunnan and Guizhou provinces in which the manure P loading was 20–30 kg P ha^{-1} (Fig. 5a, b). As economic development accelerated on the east coastal region in the 2000s and 2010s, livestock breeding developed dramatically in this region (Fig. 5c, d). The average loading of manure P in Hunan, Guangdong and Fujian provinces was about 30–40 kg P ha^{-1}. In Beijing, livestock manure P loading in arable land was even higher than 40 kg P ha^{-1}. Clearly, the regions with higher manure P loading are facing the highest environmental risk of P losses arising from livestock manure.

Due to the low environmental awareness of farmers and the lack of national waste discharge standards, there are no or inadequate disposal and treatment facilities (such as machines used for direct manure injection or incorporation into the soil to reduce surface P losses and urine—faeces segregation facilities) on 90 % of Chinese livestock farms resulting in very large amounts of manure P lost to the environment (Zheng et al. 2013; Wang et al. 2011; Sun et al. 2012). Increasing labor costs and the relatively low price of mineral P fertilizers through government subsidies encourage farmers to use mineral fertilizers instead of collecting and recycling livestock manures. This has gradually resulted in a situation in which crop and livestock production systems are no longer integrated and interdependent (Morse 1995). Consequently, livestock manures are no longer in demand for their fertilizer value, and these results in a decreasing utilization rate of manures, currently only 44 % (Cao 2006). This value is much lower than the average value of 89 % in the Netherlands (Smit et al. 2010). It was estimated that the proportion of animal wastes directly discharged to water bodies was about

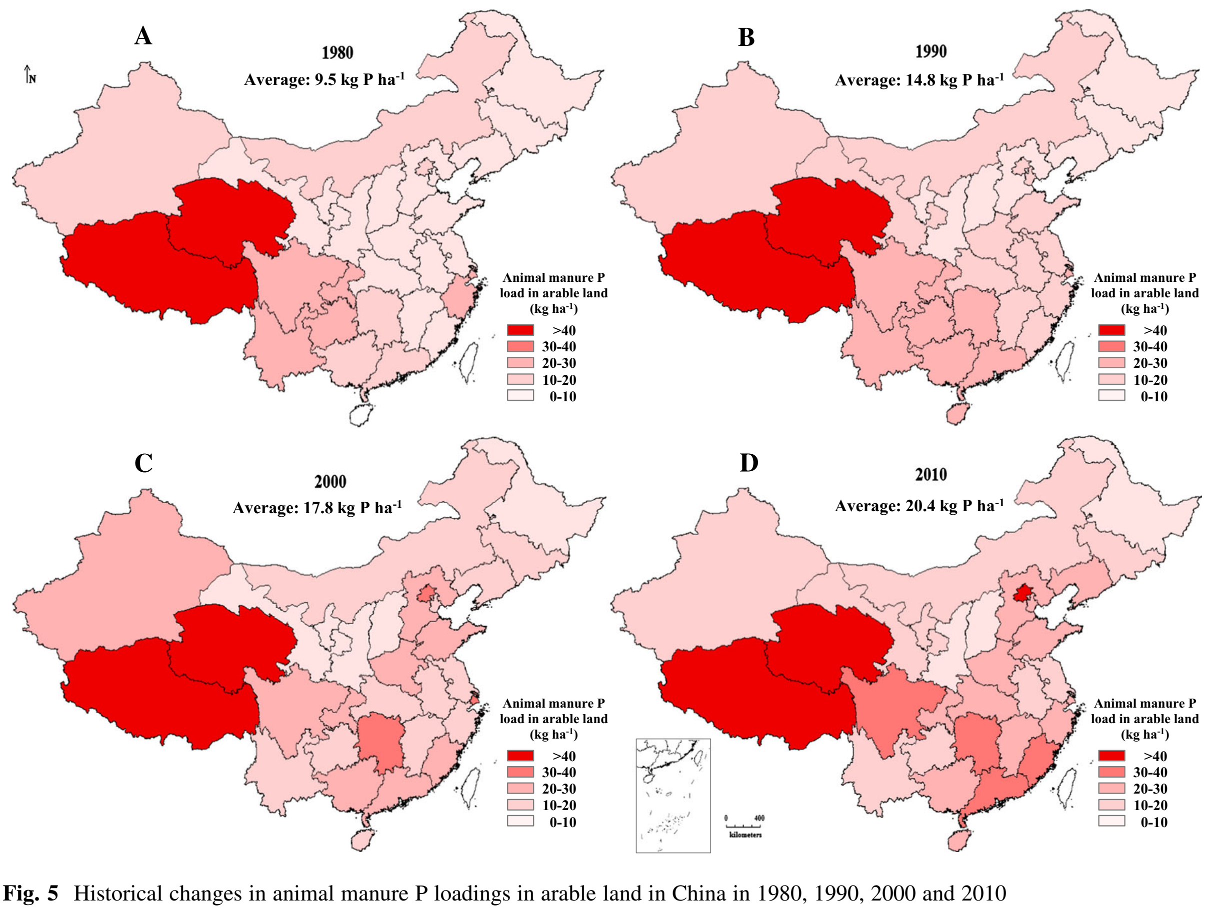

Fig. 5 Historical changes in animal manure P loadings in arable land in China in 1980, 1990, 2000 and 2010

2–8 % for solid wastes and about 50 % for liquid wastes in 2002 in China (ECCEY 2003). Not surprisingly, about 75 % of the lakes are polluted due to manure P losses and soil P erosion (Jin et al. 2005). With increasing urbanization, increasing numbers of livestock enterprises are developing around population centers and major markets where there is limited availability of arable land for manure disposal, thus exacerbating the risk of manure P losses (Zheng et al. 2013; Giller et al. 2002; Burton and Turner 2003). The P discharged from livestock manure and human excreta was larger than that from mineral P fertilizer applications and was the main contributor to environmental pollution in China in 2007, accounting for 56 % of P discharges (MEP 2010; Sun et al. 2012). Thus, improved management of livestock manures in China is an urgent task at present. Reducing excess P in animal diets and increasing the recycling of P in manure and organic wastes are the potential alternatives for a better utilization of P resources.

Soil P surplus

As demand increases for food production, Chinese agriculture is facing tremendous challenges to feed 22 % of the global population (1.3 billion) with <7 % of the global arable land. Seen as a cheap insurance policy due to the massive fertilizer subsidies from government, farmers tend to overuse mineral P fertilizers to ensure high productivity. Chinese farmers have successfully managed to increase the total amount of cereal production from 305 Mt in 1978 to 531 Mt in 2009. To achieve this, however, large amounts of mineral P fertilizers have been applied to arable land as shown in Fig. 2. Moreover, P inputs from other sources such as recycling of manure and waste P resulted in total P inputs to arable land substantially exceeding the quantity of P in harvested crops, leading to large surpluses of P in soils.

Due to soil P surplus, fertilizer use efficiency has been decreasing since 1951. The recovery ratio of P input by crops has decreased from 133 % in 1951–1955 to 44 % in 2006–2010 (Fig. 6). During 1951–1960, the balance of soil P inputs minus P outputs in agriculture was negative resulting in soil P depletion during this period, implying a PUE of more than 100 % (Fig. 6). With a dramatic increase in total P inputs compared with a slowly rising total P output since 1961, more and more P has accumulated in soils. During 2006–2010 about 16.3 Mt P accumulated in soils. It has been estimated that the total soil P surplus in arable land between 1951 and 2010 was up to 64 Mt. Many studies have also identified the substantial accumulation of soil P. Li et al. (2011) estimated that an average of 242 kg P ha^{-1} accumulated in soils from 1980 to 2007, resulting in the average soil Olsen-*P* value increasing from 7.4 to 24.7 mg P kg^{-1}. It was also reported that the average phosphate content in soils has nearly tripled between 1980 and today (Qiu 2010).

While some soil P surplus may be generally desirable from an agricultural perspective, the increased accumulation of soil P in arable land is closely related to freshwater eutrophication. Clearly, substantial P surpluses in soil result in an increased risk of P losses in surface and subsurface runoff from agricultural soils to the environment. The national survey of pollution sources in 2010 showed that the total amount of P losses from cropland was about 108,000 t, and is the second highest contributor to non-point environmental pollution (MEP 2010). This environmental problem in China is perhaps similar to or even more severe than that in Europe 10–20 years ago (Barberis et al. 1995; Djodjic et al. 2005). An evaluation of environmental sustainability among 142 countries demonstrated that China is facing greater environmental challenges than other major countries due to environmental degradation (Liu and Diamond 2005). As illustrated in Fig. 6, the average PUE in arable land in China is as low as about 44 % due to excessive P fertilization, much lower than the average values of 64 % in the European Union (Ott and Rechberger 2012) and 62 % in the United States (Suh and Yee 2011). With still increasing P fertilizer application and soil P surplus in arable soils, the contribution of P from non-point agricultural sources is predicted to increase. Therefore, a balanced fertilization for a higher PUE in cropping systems and new fertilization techniques to reduce soil P losses is potential options for a more sustainable agriculture.

Wastewater P

China's urbanization has increased from 23.0 % in 1984 to 47.5 % in 2010, driven by economic growth since its reform and opening-up to the outside world.

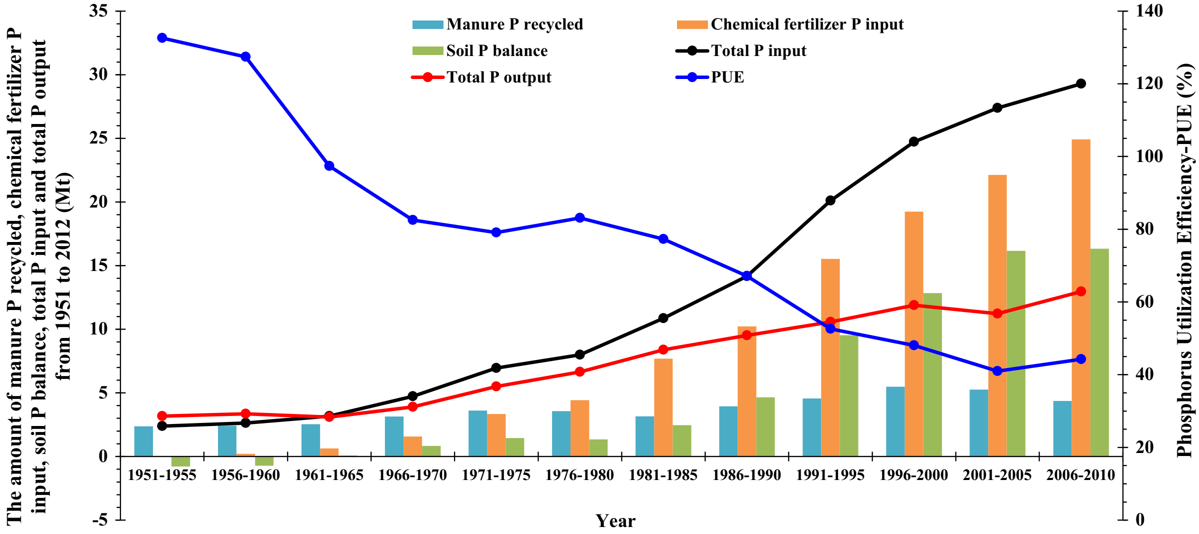

Fig. 6 Past trends in the amount of manure P recycled, mineral fertilizer P input, soil P balance, total P input, total P output and PUE in crop production during the period 1951–2010 in China. PUE was calculated by Eq. 5 in "Materials and methods" section

The net increase in the urban population has inevitably led to a high demand on water resources and an increase in municipal wastewater production. The total amount of wastewater discharged has increased from 37.3 billion tonnes (with 15.1 billion tonnes from municipal sources) in 1995 to 68.5 billion tonnes (with 46.3 billion tonnes from municipal sources) in 2012 (ECCEY 1995, 2012). However, the development of wastewater treatment plants has not kept pace with the urban sprawl in most cities. It was reported that about 14 % of urban sewage was treated whereas the remaining 86 % of urban sewage discharged into rivers and lakes without adequate treatment in 1997 (Oyang and Wang 2000). At present, although the number of urban sewage treatment plants has increased rapidly, a very large amount of sewage is still discharged without treatment. Generally, municipal wastewaters may contain 5–20 mg l^{-1} of total P, of which 1–5 mg l^{-1} is organic and the remainder inorganic, both of which are key sources of nutrients stimulating the rapid growth of marine algae (eutrophication).

Large quantities of wastewater P (ca. 0.49 Mt) were discharged in 2012 without any treatment (Fig. 7). In the populous provinces Shandong, Henan and Hebei the total amounts of wastewater P discharged to the environment were 61, 48 and 39 kilotonnes (kt) per year, respectively. Intermediate levels of wastewater P discharge occurred in Liaoning, Sichuan, Guangdong, Hunan, Heilongjiang, Hubei, Inner Mongolia and Anhui provinces where the average loadings ranged from 20 to 30 kt per year. At present there are at least three factors affecting the claimed treatment capacity in China. Firstly, there are too few wastewater treatment facilities in most cities and almost none in rural areas, resulting in the discharge of untreated sewage into the environment. Secondly, because of inefficient wastewater treatment technologies, the wastewater is not of a satisfactory standard after treatment. Thirdly, flushing toilets in urban areas mix human excreta with other (industrial) wastewater streams containing high levels of heavy metals and other toxic wastes and this greatly reduces the feasibility of nutrient recovery. A combination of these factors leads to a low wastewater P recycling ratio of about 70 % which is much lower than the average value of 90 % in the European Union (Ott and Rechberger 2012). Thus, the recovery of P from municipal and other wastewater by new technologies and developing new sanitation service infrastructure are the major opportunities for increasing the life expectancy of P resource.

Outlook on future management strategies

Based on the understanding of P intersectoral interactions along with the P flow chain, it is possible to

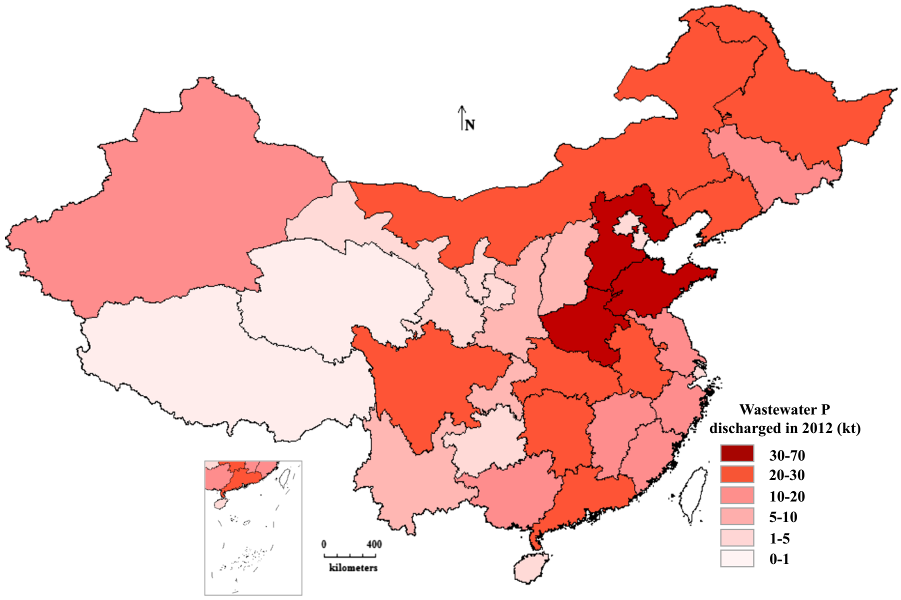

Fig. 7 The amount of wastewater P discharged to the environment in China in 2012. Data from the Chinese Ministry of Environmental Protection

develop targeted management strategies to diminish P losses to the environment and to increase P utilization efficiency (Fig. 8).

Strategy 1 for the industrial sector: phosphate rock reserves and P fertilizer production

1. New technologies at the mining stage

The quality of remaining phosphate rock around the world is declining with the depletion of high grade phosphate rock. The average grade of mined phosphate rock has declined from 15 % P in the 1970s to <13 % P in 1996 (Stewart et al. 2005; Cordell et al. 2009). The costs of mineral P fertilizer production will increase over time. How to use the remaining high grade phosphate rock more efficiently is a major challenge for the industry. New technologies may be the key to solve this problem. For instance, the average recovery rate of phosphate rock in small mining enterprises with simple picking and shoveling equipment, which select high-grade phosphate rock only, are usually below 30 % in China. This inevitably leads to a serious waste of finite P resources. In contrast, in large and medium sized mining enterprises with advanced mining equipment and modern production management technologies, the average recovery rate is up to 82 % taking open-pit and underground mines into account (Ma et al. 2012; Zhang et al. 2008). Thus, adopting new technologies for phosphate rock mining, processing and ore dressing is a crucial step to improve the PUE throughout the entire food chain.

2. Better management of by-products

Developing new technologies to promote the comprehensive utilization of the by-products including PG and low-grade phosphate rock, and prevent P losses, is a potential solution. For example, about 19 Mt of PG accounting for 27 % of annual production in China is used to make cement retarder, construction gypsum powder and gypsum brick and soil conditioner. At the same time, building impermeable bases at the storage sites of PG and low-grade phosphate rock is an efficient way to prevent contamination by leachate from diffusion into the groundwater.

3. Developing new P fertilizer products

The majority of P fertilizers applied to arable land are adsorbed by soil particles or lost in runoff after rainfall because of the high concentrations of water-soluble P in high-grade fertilizers and this finally decreases the P utilization ratio to as low as

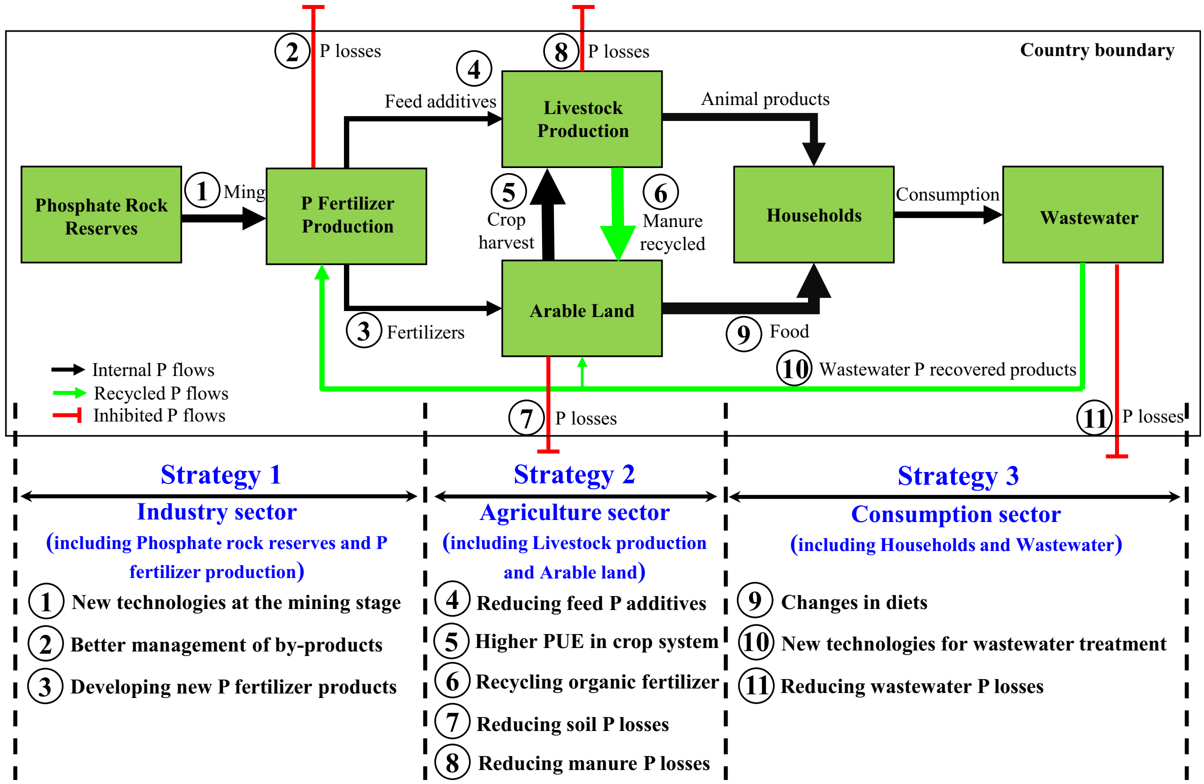

Fig. 8 Illustration of P flows and P management strategies at different parts of the P flow chain

10–20 %. Therefore, developing new P fertilizer products such as slow (controlled)-release exclusive prescriptive fertilizer (LREPF) may be an efficient way to lower fertilizer use and reduce loss of P resources. It was reported that the application of LREPF can reduce chemical fertilizer pollution, save mineral fertilizers by 20 % and raise fertilizer efficiency from 35 to 60–70 % (Qi and Gao 2008).

Strategy 2 for the agricultural sector: livestock production and arable land

4. Reducing mineral feed P additives

Phosphorus is an essential component of feed required by pigs, poultry and cattle. Therefore, adequate dietary P supply is important to meet animal daily requirements in order to maintain P homeostasis (Schröder et al. 1996). Large amounts of mineral feed P additives, consuming 9.0 % of phosphate rock in China, are commonly used in the livestock industry, usually resulting in an excess of P in the diets. It was reported that as total P in animal manure increases from the increase in dietary P, so does the proportion of inorganic P (Dou et al. 2002). Thus, reducing mineral feed P additives is a critical step to decrease manure P excretion, which in turn will minimize subsequent P losses to the environment. Addition of enzyme additives such as microbial phytases to nonruminant (pig/poultry) diets is currently considered to be an alternative mineral feed P additive to enhance the efficiency of P recovery from phytin in grain feed. Numerous animal experiments have also indicated that adding phytase additives to animal diets at 500–1000 units (FTU) kg^{-1} may replace mineral P additives for pigs and poultry and reduce their P excretion by approximately 50 % (Augspurger et al. 2003; Lei and Stahl 2001; Kim et al. 2006).

5. Higher PUE in cropping systems

First, we must recognize the importance of balanced fertilization and take it as a basic principle in cropping systems. The build-up and maintenance approach which was fully developed by Zhang et al. (2008) has successfully maintained soil Olsen-*P* at the optimum level for maximum economic efficiency and lower environmental risk of P losses (Li et al. 2011). Secondly, the activation and utilization of the 'P legacies' in soils by crops is a critical step to increase the PUE in cropping systems because fertilizer P continues to react with soil particles after application as a result of which added P gradually becomes unavailable to plants (Khasawneh et al. 1980; Sattari et al. 2012). Plant breeding and genetic modification to introduce new properties provides an effective shortcut to increase the uptake of unavailable P in soils. For example, both morphological adaptation (such as an altered root hair structure and higher fungal colonization) and physiological adaptation (such as an enhanced ability of roots to release protons, exudation of carboxylates and secretion of phosphatases) by plant breeding and genetic modification may significantly increase the PUE in cropping systems. Adjustment of cropping structure is also an efficient way to increase the PUE in cropping systems. For instance, in maize/faba bean intercropping systems the intercropped maize and faba bean had a 28–29 % higher P uptake compared with monocropped plants which has been attributed to rhizosphere interactions between intercropped maize and faba bean (Li et al. 2003).

6. Recycling organic fertilizers

The percentage of manure recycled has decreased from 100 % in 1949 to <50 % in 2005 because of the increasing application of readily available cheap and subsidized mineral fertilizers and the dramatic development of intensive animal feeding operations without enough cropland for the disposal of livestock manures (Ju et al. 2005). Thus, in agricultural production systems the recycling of organic fertilizers such as livestock manures, crop residues, household residues and other organic products should be the general norm in the future. To increase the recycling rate of organic fertilizers current financial subsidies for mineral fertilizers should be switched to organic fertilizers. At the same time, new recycling technologies for the more efficient use of organic fertilizers should also be developed. Meanwhile, a closer intensive integration of crop production and livestock systems is considered as an alternative to improve productivity, which could occur within a farm or among farms. In some regions, a relative proportion of farm land should be converted from annual cropping to rotations that include perennial forages.

7. Reducing soil P losses

In many areas excessive soil P surplus has enriched surface runoff with P due to too high a P fertilizer input. Thus, controlling the transport of soil P and preventing soil P losses to water bodies after the application of P fertilizers and manures are needed to relieve the environmental pressure caused by eutrophication.

Band-placement or localized application of P fertilizers rather than broadcasting can increase the PUE in an intensive farming system in north China and also reduce significantly soil P surface runoff (Jing et al. 2010). Other methods such as minimum- and zero-tillage, buffer zones and constructed wetlands are also useful in the minimization of soil P losses.

8. Reducing manure P losses

'Best management practices' (BMPs) for livestock manure used as fertilizer including manure collection, treatment, storage, and application are necessary to reduce manure P losses because livestock manure losses have become one of the largest contributors to runoff pollution in China.

For manure collection technologies, solid waste is collected manually and liquid waste flows along canals or pipes. The separate collection of feces and urine is important to avoid generating a large volume of dilute wastes. The solid and liquid wastes are stored in bulk in concrete/lined tanks for months before application to land and the relative importance for manure management is 20–30 % (Oenema et al. 2007). Composting is a management tool to increase the P concentration and reduce the volume of manure in an efficient way to decrease the long distance transport costs of livestock manures. Many livestock farms also use organic materials such as straw and rice hull on the ground to absorb feces and urine, which can reduce manure P losses and also provide clean and comfortable breeding houses. Commercially available chemical amendments such as alum and slaked lime are sometimes added to manure and litter to decrease P solubility and this is also a potential way to reduce the dissolved P concentration in surface runoff (Moore et al. 2000). For manure application incorporation or subsurface injection into the soil based on the local crop P requirements can significantly reduce surface P losses (Sharpley et al. 2007).

Strategy 3 for the consumption sector: households and wastewater

9. Changes in diets

The dramatic economic development and urbanization in past decades in China has promoted a meat-based diet and this is a major driver for changes in food production systems and for efficiency improvement throughout the food chain (Bai et al. 2014; Hou et al. 2013; Ma et al. 2013; Metson et al. 2012). However, through calculating backwards from human excreta to mineral P fertilizer application in arable land, it has been demonstrated that a meat-based diet per person and year requires about 11.8 kg of phosphate rock which is significantly more than the 4.2 kg requirement of a vegetarian diet (Cordell et al. 2009). Ma et al. (2013) found, based on the Chinese food dietary guidelines, that a dietary change from more meat-based diets to more cereal-based diets can significantly reduce the requirement for arable land, water, mineral P fertilizer and reduce P losses. Metson et al. (2012) also reported that meat consumption is the largest contributor to 'dietary P footprints'. Therefore, decreasing future meat consumption could play an important role in sustainable P management strategies. Though changing dietary habits is a great challenge which is related closely to increased urbanization and higher incomes, a dietary change toward less meat and a reduction of milk consumption versus more vegetables and cereals is considered a promising way to reduce wastage of P resources.

10. New technologies for wastewater treatment and (11) reducing wastewater P losses

A large quantity of P in discharged wastewater with more than 0.03 mg P l^{-1} is one of the main causes of eutrophication that has exerted great pressure on the environment in China (Gachter and Imboden 1985). Currently, urban wastewater infrastructure is basically designed for public health but is increasingly focusing on P recovery from both urban and industrial wastewaters (Reindl 2007). Total removal or at least a significant reduction in P concentrations in wastewaters is now a legal obligation in many countries.

According to the types of wastewater and operational costs, many different P-removal technologies are available to treat wastewaters. Based on the different mechanisms for removing P, the technologies include chemical metal precipitation, absorbing materials, physical heat treatment and biological immobilization by bacteria, microalgae and plants (De-Bashan and Bashan 2004). These processes essentially transfer P from the liquid to the sludge phase. The P removal efficiency of these technologies ranges from 39 % to 100 %. For example, some metal oxides such as iron oxide, $Al(OH)_3$, calcite, and $Mg(OH)_2$ are often used as reagents in metal precipitation technology to react chemically with P in wastewater, generating insoluble crystalline compounds (Baker et al. 1998; Donnert and Salecker 1999; Moriyama et al. 2001; Shin and Lee 1998; Wu et al. 2001). Depending on the characteristics of the P removal technologies being used, the P recovered from wastewaters can be reused either directly as a fertilizer applied to arable land or as a raw material for the fertilizer industry. Phosphorus recovery from wastewaters is not only a sustainable way to prevent P losses but also provides an alternative source of phosphate fertilizer for agricultural production. Meanwhile, more stringent legislation to reduce the concentration of P in municipal wastewaters before discharge into the environment is an efficient way to reduce wastewater P losses. Reuse of residential wastewaters composed mainly of human urine and fecal matter to irrigate crops is also an alternative way to reduce wastewater P losses. It was reported that more than 25 % of urban vegetables are being fertilized with wastewaters from cities (Ensink et al. 2004). However, this direct reuse of wastewaters requires a sanitation service infrastructure that avoids the mixing of human excreta with other wastewater streams containing heavy metals and other toxic wastes.

If all of the strategies described above were adopted in P management in China, PUE in different parts of the P flow chain would be increased significantly and large amounts of P resources would be saved (Table 1). We reviewed most of the P flow studies at national scale around the world, and determined the maximum PUEs in different sectors, which were considered as the attainable PUEs in China (Table 1). For example, the maximum recycling ratio of manure P reported in literature was about 89 % in the Netherlands (Smit et al. 2010). Thus, the increase in recycling ratio of manure P from 44 to 89 % in China is actually possible if the manure management technologies could achieve the standards of the Netherlands. Based on the PUEs and the size of P flows in different parts in China in 2010, the optimized size of P flows and the amount of P resource saved after adopting technologies were determined.

If in P fertilizer production new technologies are introduced at the phosphate rock mining stage, the PUE will increase from 61 % to the attainable value of 98 % as in the United States. This would not be hard to achieve by investing massively in the advanced mining equipment and staff skills training. If the mineral P fertilizer production in 2010 in China is continued, about 4.3 Mt P in phosphate rock will be saved which equals to 32.9 Mt phosphate rock (average P_2O_5 content = 30 %). In arable land production, after adopting new technologies the PUE may increase from 44 % to the attainable value of 64 % in the European Union and about 2.5 Mt P fertilizers will be saved to produce the same crop yield as in 2010. This is eminently possible if China invests in a set of integrated soil-crop management practices based on a modern understanding of crop ecophysiology and soil biogeochemistry and makes it viable for farmers (Chen et al. 2014). If new technologies are applied in manure and wastewater P management, about 1.6 and 0.3 Mt P will be recycled and used as P fertilizer sources, respectively. An investment in the municipal and sanitation service infrastructure is essential for the recovery of P from wastewater, especially in rural areas. However, manure P recycling through incorporation or subsurface injection with large machines on 100s of millions of small parcels of land will be a huge challenge in China.

The attainable maximum PUE in the whole chain was about 21 % in South Korea (Jeong et al. 2009), which is much higher than the 6 % in China. If the PUE in the whole chain could increase from 6 to 21 % with new technologies, about 8.1 Mt P resources will be saved in China (Table 1). In order to achieve this, a concerted effort by all entities along the whole P flow chain towards efficient use of P resource is needed in China.

The way forward

Phosphorus is a finite, yet essential, resource for agriculture and thus food production and China is the world's largest consumer (30 % on just 7 % of the world's arable land) and an important producer (37.5 % of the global total). In this paper we identified China's P fertilizer industry, soil P surplus, livestock manure P and wastewater P recycling as the priority sectors that need attention to mitigate P use and thereby improve PUE. China can learn from other countries with intensive agriculture that have been dealing with large nutrient surpluses (i.e. in Europe) such as innovation of technology and policy to reduce over-application of P fertilizers in intensive agriculture by at least 30 %, in vegetable, fruit and high value cash crop production systems. China also needs the appropriate incentives and regulations from the government to promote the large-scale and effective re-use of P resources from all wastes and by-products. Most important is to develop a novel knowledge transfer and technology extension system for millions of small-holder farmers in order to realize the new governmental policies of zero-increase in chemical fertilizer input in crop production by 2020 and

Table 1 Comparison of PUEs and P situations before and after adoption of new technologies in China in 2010

Different sectors	PUEs in 2010 (%)	Attainable PUEs after adopting technologies (%)	Size of P flows in 2010	Optimized size of P flows in 2010 if adopting technologies	Amount of P resource saved (Mt P)
PUE in P fertilizer production	61	98 (Zhang et al. 2008)	11.3 Mt P mined	7.0 Mt P demanded	4.3
PUE in arable land production	44	64 (Ott and Rechberger 2012)	8.1 Mt P input	5.6 Mt P demanded	2.5
Manure P recycling ratio	44	89 (Smit et al. 2010)	1.6 Mt P recycled	3.2 Mt P recycled	1.6
Wastewater P recycling ratio	70	90 (Ott and Rechberger 2012)	0.5 Mt P discharged	0.2 Mt P discharged	0.3
PUE in the whole chain[a]	6	21 (Jeong et al. 2009)	11.3 Mt P mined	3.2 Mt P demanded	8.1

[a] PUE in the whole chain = P in food digested by human/phosphate rock mined

transformation of high input and high output agriculture to efficient and sustainable intensification. This requires an integrated effort of scientists, policy makers, farmers and other stakeholders by working together and inducing fundamental changes in P utilization.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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