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The Human Intensified Global Phosphorus Flows and Environmental Impacts

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Contents

Abstract

Human activities have significantly intensified natural phosphorus cycles, which resulted in some serious environmental problems that modern societies are facing today. This paper attempts to quantify global phosphorus fluxes associated with present mining, farming, animal feeding and household consumption. Varieties of physical characteristics of the related phosphorus fluxes as well as their environmental impacts in different economies, including the United State, European countries and China, are taken into consideration. Particular attentions are given to the global phosphorus budget in cropland and the movement and transformation of phosphorus in soil, because these phosphorus flows in relation with the farming sector constitute major fluxes that dominate the anthropogenic phosphorus cycle. The results show that the global input of phosphorus to cropland, in both inorganic and organic forms from various sources, cannot compensate the removal in harvests and the losses of erosion and runoff. A net loss of phosphorus from the world's cropland is estimated at about 10.5 MMT P each year, nearly one half of the applied phosphorus.

Keywords: phosphorus cycle, phosphate fertilizer, eutrophication, soil chemistry, and industrial ecology.

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1. The human intensified phosphorus cycles

Inorganic cycle

Phosphorus circulates through the environment in three natural cycles. The first of these is the inorganic cycle, which refers to phosphorus in the crust of the Earth. Through millions of years, phosphorus has moved slowly through the inorganic cycle, starting with the rocks which slowly weather to form soil, from which the phosphorus is gradually leached from the land into rivers and onward to the sea, where it eventually forms insoluble calcium phosphate and sinks to the sea floor as sediment (Follmi 1996). There it remains until it is converted to new, so-called sedimentary rocks as a result of geological pressure. On a time scale of hundreds of millions of years, these sediments are uplifted to form new dry land and the rocks are subject to weathering, completing the global cycle (Schlesinger 1991). In addition, some phosphorus can be transferred back from the ocean to the land by fish-eating birds whose droppings have built up sizeable deposits of phosphate as guano on Pacific coastal regions and islands, and by ocean currents that convey phosphorus from the seawater to these regions. A simplified schematic of the global phosphorus cycle is presented as Figure 1.

The global cycle of phosphorus is unique among the cycles of the major biogeochemical elements in having no significant gaseous compounds. The biospheric phosphorus flows have no atmospheric link from ocean to land. A little phosphorus does get into the atmosphere as dust or sea spray, accounting for 4.3 million metric tons of phosphorus per year (MMT P/yr) and 0.3 MMT P/yr (Richey 1983), respectively, but the amounts are several orders less important than other transfers in the global phosphorus cycle. The amount of 4.6 MMT P/yr of atmospheric phosphorus deposition, being balanced by the phosphorus carried by the wind and the seaspray, cannot offset the endless drain of this element from the land due to erosion and river transportation (Schlesinger 1991). Fortunately, increased anthropogenic mobilization of the element has no direct atmospheric consequences.
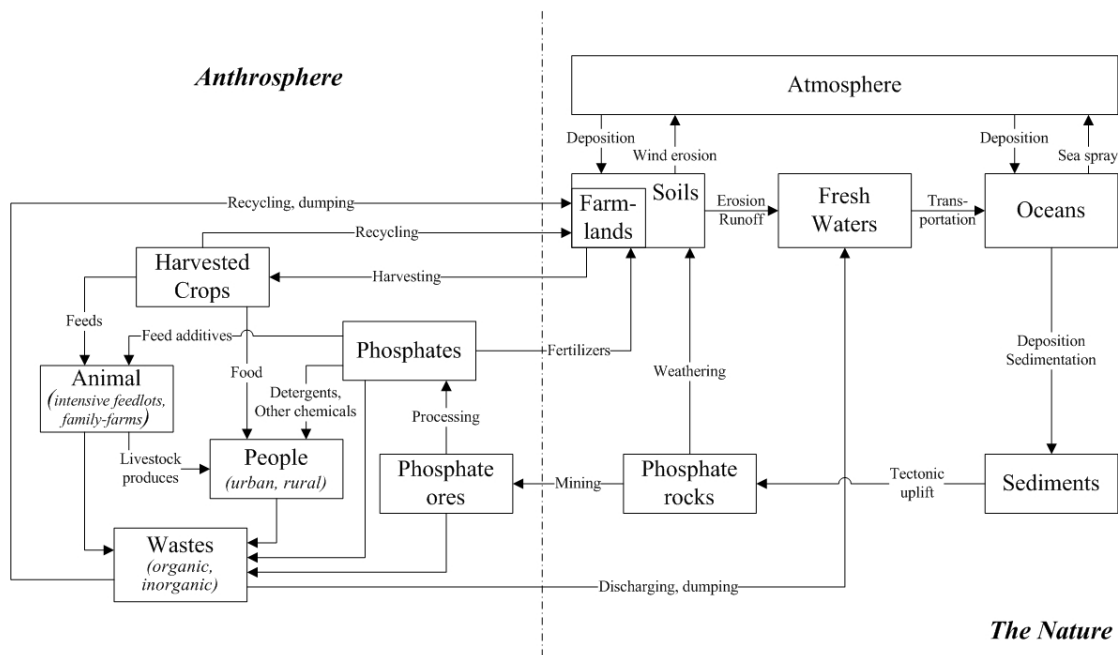
Figure 1. The human intensified global phosphorus cycles

Sources: modified from (Liu, Chen, and Mol 2004; Richey 1983; Smil 2000).

Nearly all the phosphorus on land is originally derived from the weathering of calcium phosphate minerals, especially apatite [$Ca_5(PO_4)_3OH$]. Around 13 MMT P/yr of this is released to form soils each year (Emsley 2000). However, this amount cannot offset the annual losses of phosphorus from the land. Taking into account all four forms of phosphorus (dissolved and particulate, organic and inorganic), the total amount of annual phosphorus losses from the lithosphere into freshwaters is estimated at 18.7–31.4 MMT P/yr (Compton et al. 2000).

The uncertainty in the estimate is mainly due to a lack of knowledge on the biogeochemical processes of the particulate inorganic phosphorus (PIP), which constitutes the major component in the total loss. Not all of the eroding phosphorus can eventually reach the ocean. About 3.0 MMT P/yr carried away by wind into atmosphere, and at least 25% of that is re-deposited on adjacent cropland and grassland or on more distant alluvia (Smil 2000). Consequently, the amount of phosphorus transported by freshwaters into the ocean is probably in the range 12–21 MMT P/yr. This result agrees with the most likely value of 17–22 MMT P/yr given by some previous estimates (Emsley 1980; Howarth et al. 1995; Meybeck 1982; Richey 1983; Sposito 1989). The riverborne transport of phosphorus constitutes the main flux of the global phosphorus cycle. The loss, as a result of erosion, pollution and fertilizer runoff, must be considerably higher than it was in prehuman times. It can be argued that the human

intensified phosphorus flux caused by wind and water erosion is at least double (Compton et al. 2000), or even three times (Smil 2000) its prehistoric level.[1]

Organic cycles

Imposed on the inorganic cycle are two organic cycles which move phosphorus through living organisms as part of the food chain. These are a land-based phosphorus cycle which transfers it from soil to plants, to animals and back to soil again; and a water-based organic cycle which circulates it among the creatures living in rivers, lakes and seas. The land-based cycle takes a year on average and the water-based cycle organic cycle only weeks. It is the amount of phosphorus in these two cycles that governs the biomass of living forms that land and sea can sustain.

The amount of phosphorus in the world's soils is roughly 90×10^3–200×10^3 MMT P according to various estimates (Emsley 1980; Filippelli 2002; Meybeck 1982; Richey 1983). While the total phosphorus content of soils is large, only a small fraction is available to biota in most soils. This constitutes an available phosphorus pool containing 1,805–3,000 MMT P (Grove 1992), most likely 2,000–2,600 MMT P (Emsley 1980; Richey 1983). A larger amount, in the range 27×10^6–840×10^6 MMT P, can be found in the oceans. The sea water contains 80×10^3–120×10^3 MMT P and the rest is accumulated in sediments (Emsley 1980; Filippelli 2002; Grove 1992; Richey 1983; Smil 2002).

The ocean water loses phosphorus continually in a steady drizzle of detritus to the bottom, where it builds up in the sediments as insoluble calcium phosphate. Despite the geological remobilization, there is a net annual loss of millions of tons of phosphate a year from the marine biosphere. Thus the ocean sediments are by far the largest stock in the biogeochemical cycles of phosphorus.

Mineral reserves and extraction

Humans have disturbed and enhanced the global phosphorus cycles, since phosphate rock was firstly intensively exploited in the late-1860s in North Carolina, the United States. Most of the economic deposits of phosphate rock have subsequently been found in Morocco and the Western Sahara, China, USA (mainly in Florida, Tennesee and North Carolina), Australia and some other countries. The world phosphate reserves that can be economically extracted or produced near current prices is given as 2,400 MMT P, or 5,400 MMT P_2O_5 (USGS 2006). The reserve base, representing the part of identified resource which meets specified minimum physical and chemical criteria related to current mining and production practices, is estimated as 6,500 MMT P (15,000 MMT P_2O_5) in total (USGS 2006). In 2005, the world extraction of phosphate rock was 19.4

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[1] Some authors didn't. Schlesinger (1991), for instance, assumed that this flux may be only slightly higher than in prehistoric time.

MMT P (44.4 MMT P_2O_5), roughly equivalent to the amount of phosphorus transported by rivers to the ocean.

Global phosphate consumption

The phosphate rock is initially converted to phosphoric acid (P_2O_5) by reaction with sulfuric acid. The phosphoric acid is further processed to produce fertilizers, food-grade and feed-grade additives and detergents. Other marginal applications include metal surface treatment, corrosion inhibition, flame retardants, water treatment and ceramic production. Despite such widespread use, the latter applications represented only about 3 per cent of the total consumption of various phosphates in the 1990s (CEEP 1997).

The global consumption of all phosphate fertilizers surpassed 1 MMT P/yr during the late-1930s. After reaching 14 MMT P/yr in 1980, the world consumption of phosphate fertilizers has been relatively stable. It was 14.8 MMT P (34 MMT P_2O_5) in statistical year 2002/2003, and slightly decreased to 13.8 MMT P (31.5 MMT P_2O_5) in statistical year 2003/2004 (FAO 2005), roughly accounting for 72.8% of the global extraction of phosphate rock. As the top fertilizer-processing country, China contributed 22% and 28%, respectively, to the world production and consumption; the top three economies (China, United States, and India) accounted for one half of the world consumption. The area of the world current crop land is about 1.4 billion hectare (FAO 2005), implying the global fertilizer application intensity averages 10 kg P/ha. The application rate varies significantly among continents, ranging from about 3 kg P/ha in Africa to over 25 kg P/ha in Europe. Among Western Europe countries, application levels range from 8.7 kg P/ha in Denmark and Sweden to 34 kg P/ha in Ireland (Johnston and Steen 2000).

Crop harvests

The use of phosphates to nourish agricultural soils aims to replenish the removal of phosphorus from soil by harvests and erosion losses. Adopting the average phosphorus contents in crops and the harvest index (Smil 1999), the global crop production harvested 12.7 MMT P from soils in 2005 as shown in Table 1, based on the world agricultural production database (FAO 2006a). A study of Chinese phosphorus flows suggested that the national harvest in 2000 removed 3.4 MMT P from croplands, based on a set of 'domestic' data for the phosphorus contents and the harvest index (Liu 2005). These two estimates agree that: (1) cereals accounted for a major part of the harvested phosphorus, i.e. 70% at the global level, and/or 68% in China; and (2) about two thirds of the annually harvested phosphorus is contained in grains, and rest is contained in straw and other agricultural waste.

Table 1. Allocation of phosphorus in world harvest in 2005

	Harvested crops			Crop residues		Total P uptake (MMT P)
	Fresh weight (MMT)	Dry matter (MMT)	P in grains (MMT P)	Residues (MMT)	P in straws (MMT P)	
Cereals	2,239	1,968	5.9	2,947	2.9	8.9
Suger crops	1,534	476	0.5	370	0.7	1.2
Roots & tubers	713	143	0.1	219	0.2	0.4
Vegetables	882	88	0.1	147	0.1	0.2
Fruit	505	76	0.1	126	0.1	0.2
Pulses	61	58	0.3	61	0.1	0.4
Oilcrops	139	102	0.1	92	0.1	0.2
Other	100	80	0.1	200	0.2	0.3
Forages		500	1.0	0	0.0	1.0
Total	**6,173**	**3,491**	**8.2**	**4,163**	**4.5**	**12.7**

Sources: harvest data are derived from (FAO 2006a); conversion coefficients are adopted from (Smil 1999).

Since natural weathering and atmospheric deposition, as discussed elsewhere, cannot compensate the amount of phosphorus uptake from soils, application of phosphates, in both inorganic and organic forms, becomes essential to sustain today's harvests. There are several means of organic phosphorus reuse. The most direct means is to recycle crop residues *in situ*. Assuming roughly half of the annual output of crop residues (mostly cereal straw) is not removed from fields, the amount of the direct reuse of crop residues is about 2.2 MMT P/yr.

Livestock and animal wastes

Animal wastes have been applied as organic manure in traditional farming and remain a relative large source of recyclable phosphorus in modern agriculture. A rough estimate of global production of animal wastes is needed, because detailed inventories of nutrient budget of livestock husbandry are not available for most countries. According to the latest national survey data from the United States, beef cattle, dairy cattle, swine and poultry produced 1.7 MMT P contained in animal manures in 1997, of which about half was produced by confined animals (Kellogg et al. 2000). The livestock population in the United States accounts for 7% of the world total in 1997 and the proportion has remained fairly constant. On this basis, the global production of animal wastes would be 24.0 MMT P/yr. However, the real figure may be somewhat smaller, because the

animals in U.S. are exceptionally well-fed. For this reason, Smil's estimate of global production of 16–20 MMT P/yr in animal wastes, obtained by applying an average concentration of 0.8–1% of phosphorus for both confined and unconfined animal wastes, is probably more accurate (Smil 2000).

Only the phosphorus in confined animal wastes is considered to be recyclable for croplands, while unconfined animal wastes mainly returns to pastures. Assuming animal biomass remains relatively constant, the amount of phosphorus in animal wastes is equal to the consumption of phosphorus contained in all kinds of feeds. According to the global Food Balance Sheet (FBS) in 2003, livestock consumed 36% of the harvested cereals (excluding the amount of cereals processed for beer), 21% of the harvested starchy roots, and 20% of the harvested pulses (FAO 2006b). Consequently, the annual livestock consumption of phosphorus in the harvests is accounted for about 2.9 MMT P/yr.

Some part of crop residues is used as animal fodder. However, the reuse ratio of crop residues as fodder varies considerably. For instance, it was reported that the percentage of crop residues – mostly the straws of rice, wheat and corn (maize) – used as fodder ranged from 3.6% in Shanghai to 42.8% in Gansu province in 2000 in China, depending on crop and livestock species, farming and feeding traditions, and local economic profiles, and averaged 22.6% across the nation (Gao et al. 2002; Han et al. 2002). Since over 70% of world livestock are raised in developing countries where commercial feeds are less used, the global recycling rate of crop straws as fodder is probably about 25%, leading to an absolute quantity of 1.1 MMT P/yr.

Another major source of animal daily phosphorus intake is via feed additives. Around 6% of the global yield of phosphoric acid has been processed as animal feed-grade additives since 2000 (Brasnett 2002; PotashCorp 2004). This constitutes an annual phosphorus flux of 1.0 MMT P/yr input to livestock husbandry.

Adding all above three sources, the global livestock consumption of phosphorus amounts to about 5.0 MMT P each year. Taking into account the recycling of various industrial byproducts, and kitchen organic wastes (which is prevalent in rural family-based farms in developing countries), this figure could be as much as 20% higher, resulting in a total of 6.0 MMT P/yr. Of course, the phosphorus flux to livestock of 6.0 MMT P/yr is mainly consumed by animals in confined facilities, while the world cultivated and natural pastures provides a major source of phosphorus for unconfined animals. Assuming 1.0 MMT P/yr goes to unconfined animals in pastures, about 5.0 MMT P/yr consumed by confined animals gives an approximation for the maximum potential of recoverable phosphorus for croplands. If one half of the organic phosphorus

in confined animal wastes is subject to recycling, animal manure is responsible for about 2.5 MMT P/yr returns to global croplands.

Food consumption and human wastes

The third source of organic phosphorus available, in principle, for cropland is human waste. Assuming the world human body mass averaging 45 kg/capita (reflecting a higher proportion of children in the total population of low-income countries) and phosphorus content in human body, averaging 470 g P/capita, implies a global anthropomass contains approximately 3.0 MMT P. The typical daily consumption is about 1,500 mg P/capita for adults (CEEP 1997). This is well above the dietary reference intake (DRI), the amount human individual should take in each day, as recommended by the Food and Nutrition Board, Institute of Medicine, US National Academy of Science. The US recommended intakes are 700 mg/capita for adults over 18 years of age, 1,250 mg/capita for young adults between 9 and 18 years of age, and 500 mg/capita for children.

A similar estimate for China is derived from a SFA model: the individual daily intake of phosphorus was 1,400 mg P/capita for urban residents and 1,470 mg P/capita for rural residents in 2000 (Liu 2005). This exceeds the DRI of 1,000 mg/capita recommended by the Chinese Nutrient Society. In addition, livestock products provided 30% of daily phosphorus intake for Chinese urban residents, and 14% of that for rural population (Liu 2005).

Although the phosphorus content of anthropomass is marginal in comparison with that of soil biota or ocean biota, the environmental consequences can still be significant. Giving the annual world population growth of 78.4 million since 2000 (UN 2004), the net accumulation of phosphorus in the global anthropomass is around 0.04 MMT P/yr. Compared with the global consumption of phosphorus in foods, the slight increase of phosphorus in the anthropomass stock implies a low assimilation rate of about 0.5%. Assuming a global average dietary consumption of 1,400 mg P/capita, human excreta must have contained about 3.3 MMT P/yr, of which urban and rural population generated 1.6 MMT P/yr and 1.7 MMT P/yr, respectively.

Application of human excreta as organic fertilizer is common both in Asia and in Europe, but less prevalent elsewhere the world. The nutrient linkage between farmers and croplands has been relatively stable, but the human wastes in urban areas are less recycled than in rural areas. In China, less than 30% of human wastes in urban areas were recycled for agricultural uses in the late 1990s (Chen 2002). This percentage dramatically decreased from 90% in 1980 (Chen and Tang 1998). In contrast, about 94% of human wastes in rural areas were returned to croplands in the 1990s (Pan et al. 1995). In European countries, the recycling rate of urban sewage averaged about 50%

over the 1990s (EEA 1997; Farmer 1999). Globally, it could be appropriate to assume that about 20% of urban human wastes and about 70% of rural human wastes are recycled at present. Therefore, recycled human wastes amounts to 1.5 MMT P/yr.

Adding the quantities of the phosphorus recycled as crop residues, animal manures and human wastes, the total organic fertilizers applied to croplands amounts to 6.2 MMT P/yr. This is equivalent to 45% of the applied amount of inorganic fertilizers. Thus, the global input of phosphorus to croplands is probably 20 MMT P/yr in total, or 1.6 times the amount of the phosphorus removed from the soil by harvesting. This leads to a net accumulation of 7.3 MMT P/yr or 4.7 kg P/ha in global soils, disregarding erosion and runoff losses.

Phosphates in soil
The distribution, dynamics and availability of phosphorus in soil are controlled by a combination of biological, chemical and physical processes. These processes deserve special attention, as a considerable proportion of the applied phosphate is transformed into insoluble calcium, iron or aluminum phosphates. On average only a small proportion, perhaps 15–20% of the total amount of phosphorus in the plant, comes directly from the fertilizer applied to the crop. The reminder comes from soil reserves (Johnston and Steen 2000). For most of the twentieth century, farmers in western countries were advised to add more than double the amount of phosphate required by a crop, because these immobilized calcium, iron and aluminum phosphates had been assumed to be permanently unavailable to plants.

The primary source of phosphorus taken up by plants and microorganisms is dissolved in water (soil solution). The equilibrium concentration of phosphate present in soil solution is commonly very low, below 5 μMol (Condron and Tiessen 2005). At any given time, soil water contains only about 1 per cent of the phosphorus required to sustain normal plant growth for a season (Emsley 2000). Thus, phosphates removed by plant and microbial uptake must be continually replenished from the inorganic, organic and microbial phosphorus pools in the soil. These continuous processes dominate contemporary agricultural production to remove about 8.2 kg P/ha from cropland each year on a world average (based on our own estimate), and commonly 30 kg P/ha from the US and European fertile agricultural soils (Johnston and Steen 2000).

Each phosphate mineral has a characteristic solubility under defined conditions (Burke, Heathwaite, and Preedy 2004). The solubility of many compounds is a function of acidity (pH). A typical solubility diagram illustrates phosphate solubility against pH is shown in Figure 2. An increase in pH can release sediment bound phosphorus by increasing the charge of iron and aluminum hydrous oxides and therefore increasing the competition between hydroxide and phosphate anions for sorption sites. Also, organic

acids can inhibit the crystallization of aluminum and iron hydrous oxides, reducing the rate of phosphorus occlusion (Schlesinger 1991). The production and release of oxalic acid by fungi explains their importance in maintaining and supplying phosphorus to plants.
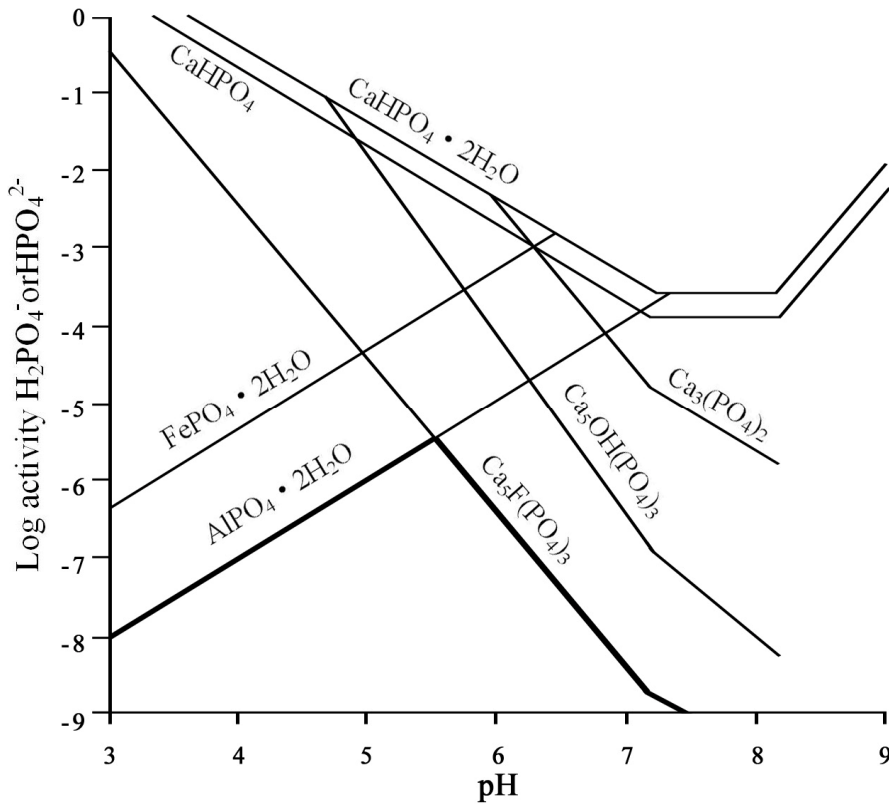

Figure 2. The solubility of phosphorus in the soil solution as a function of pH

Sources: adopted from (Schlesinger 1991).

The relative sizes of the sources and stocks of phosphate in soil change as a function of soil development as shown in Figure 3. The buildup of organic phosphates in the soil is the most dramatic change. As time goes on this becomes the chief reservoir of reserve phosphate in the soil. In most soils, organic phosphates range from 30% to 65% of total phosphorus, and it may account for as high as 90%, especially in tropical soils (Condron and Tiessen 2005).

The reasons for this are their insolubility and chemical stability. Several authors have noted that acid soils tend to accumulate more total organic phosphorus than do alkaline soils. This is almost certainly because organic phosphates react with iron and aluminum under acid conditions and become insoluble (Harrison 1987). Being the salts or metal complexes of phosphate esters they release their phosphate by hydrolysis, but only very slowly. Phosphate esters can have half lives of hydrolysis of hundreds of years. This

process can be greatly speeded up by the action of phosphatase enzymes in the soil whose function is to facilitate reaction by catalyzing it (Quiquampoix and Mousain 2005).

At later stages of soil development, phosphorus is progressively transformed into less-soluble iron- and aluminum-associated forms, and organic phosphorus contents of the soil decline (Tiessen and Stewart 1985). At this stage almost all available phosphorus is found in a biogeochemical cycle in the upper soil profile, while phosphorus in lower depths is primarily involved in geochemical reactions with secondary sediments (Schlesinger 1991).
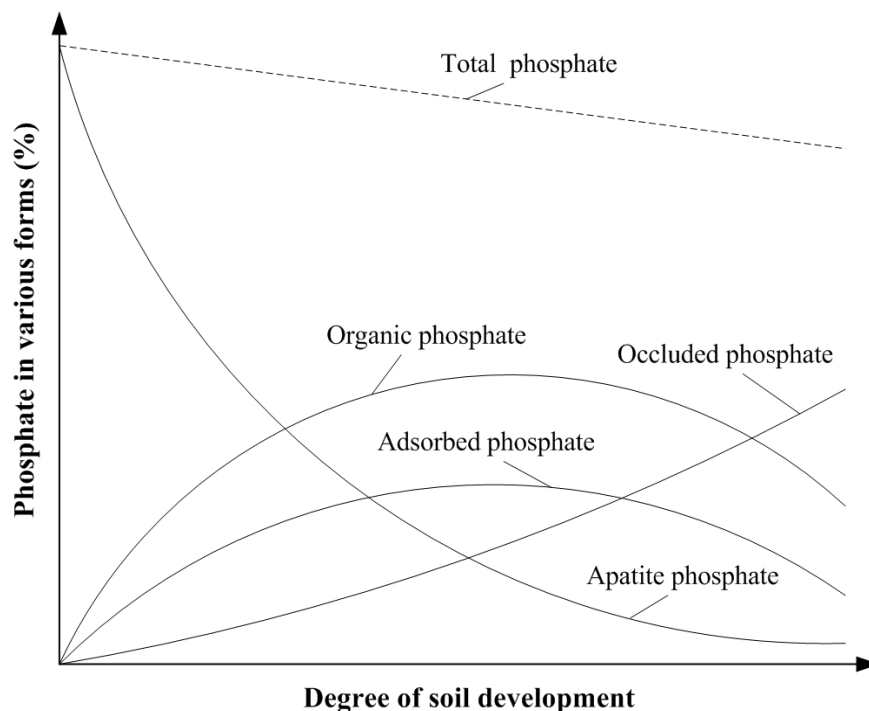

Figure 3. Phosphates in the soil vary with soil development

Sources: adopted from (Emsley 1980).

When the dissolved phosphates supply to growing biomass is abundant, a net immobilization of inorganic phosphorus into organic forms will occur. *Vice versa*, inadequate inorganic phosphorus supply will stimulate the production of phosphatases and the mineralization of labile organic forms of phosphorus for microbial uptake (Tiessen and Stewart 1985). A continuous drain on the soil phosphorus pools by cultivation and crop removal will rapidly deplete both labile inorganic and organic phosphorus in soils.

Allowing soil reserves of readily available phosphorus to fall below a critical value, determined by field experiments, can result in a loss of yield. The turnover of available

phosphates by plants in soil solution is determined by rates of releases of phosphorus from insoluble forms to soluble phosphates. All kinds of soil particles can contribute to this process and in some cases it is not only chemical balance that maintains the supply but the action of microbes and enzymes that release phosphate from organic debris in the soil. It is believed that the biogeochemical control of phosphorus availability by symbiotic fungi is a precursor to the successful establishment of plants on land (Schlesinger 1991).

However, our existing knowledge – briefly discussed above – cannot yet provide a comprehensive understanding of the complex movements and transformations of phosphorus, especially of its organic forms (Turner, Frossard, and Baldwin 2005). This hampers the efficient application of phosphate fertilizers, and the efficient control of phosphorus losses.

<u>Phosphorus losses</u>
Phosphorus is lost from croplands via erosion or runoff. Quantifying phosphorus losses in eroding agricultural soils is particularly uncertain, as erosion rates vary widely even within a single field. It is also because that few nations have comprehensive, periodic inventories of their soil erosion.

The phosphorus loss from croplands can be roughly estimated based on the amount of topsoil erosion and average phosphorus content. A crop takes up the majority of the nutrients it requires from topsoil. The topsoil is often identified as the 'plough layer', i.e. the 20–30 cm depth of soil which is turned over before seedbed preparation. The volume of topsoil in the plough layer is around 2,500 m^3 per hectare and weighs approximately 2,000 tonnes (Johnston and Steen 2000). A ton of fertile topsoil can contain 0.6–3.0 kg of phosphorus, mainly based on US and European agricultural practices (Johnston and Steen 2000; Troeh, Hobbs, and Donahue 1991).

It has been estimated that the annual soil erosion from agricultural systems of the United States, China and India was 3.0 Gt/yr, 5.5 Gt/yr and 6.6 Gt/yr, respectively (Pimentel 2006). Since these three countries hold 23% of global agricultural lands, the global soil erosion could reach 66 Gt/yr. Some the most serious soil erosion takes place in the agricultural systems of Southeast Asia, Africa and South America (Yang et al. 2003). Hence, the real global soil erosion loss could be even higher, perhaps as much as 75 Gt/yr (Myers 1992).

The erosion intensity from croplands varies a lot among countries, ranging from 0.5–400 t/ha-yr (Pimentel et al. 1995). Worldwide, soil erosion is highest in the Asia, Africa and South America, averaging 30–40 t/ha-yr of soil loss (Taddese 2001). Smil has suggested that the global average erosion rate is at least 20 t/ha-yr (Smil 2000). The

lowest erosion rates occur in the United States and Europe where they average about 10 t/ha each year (USDA 2000a; USDA 2000b). It is evident that soil erosion in the US has been reduced by soil conservation policies: a national survey showed that the total soil erosion between 1982 and 1992 decreased by 32% (Uri 2001). The annual sheet and rill erosion rate in the US fell from an average of 10 t/ha in 1982 to 7.7 t/ha in 1992, and the wind erosion rate fell from an average of 8.1 t/ha-yr to 5.9 t/ha-yr over the same period. Assuming global erosion rate averaging 25 t/ha gives the soil loss of 38.5 Gt/yr from cropland (cf. Table 1). Furthermore, most of the loss is permanent and may not be replenished by weathering. For instance, the excessive soil loss, a rate that would impair long-term crop productivity, is estimated at about 25.4 Gt/yr in around 1980 (Brown and Wolf 1984).

Erosion from pastures is commonly less intensive than that from ploughed fields. However, soil losses have been greatly increased by overgrazing, which now affects more than half (that is, at least 1,720 million hectare) of the world's permanent pastures with a high erosion rate of 15 t/ha each year (Smil 2000). This leads to 25.8 Gt/yr of the soil loss from overgrazed pastures. Together with the amount of soil loss from cultivated grassland, the world permanent pastures lose its topsoil at an annual rate of 34.4 Gt/yr. Adding the losses from cropland and pastures amounts the world soil erosion from agricultural areas to 72.9 Gt/yr in total, or 15 t/ha-yr on average. This is similar to previous estimates as discussed above.

Allowing for the poor condition of topsoil in developing countries, it might be appropriate to assume that the global phosphorus content in topsoil averages about 0.5 kg P/t, or 1.0 t P/ha. This gives the world phosphorus losses at 19.3 MMT P/yr from cropland and at 17.2 MMT P/yr from pastures, respectively, as shown in Table 2.

Table 2. Global soil erosion and phosphorus losses from agricultural land (2003)

	Cropland	Permanent pasture	
		Overgrazed	Ordinary
Total area (million he)	1,540	1,720	1,720
Soil erosion			
Erosion rate (t/ha-yr)	**25**	**15**	**5**
Erosion quantity (Gt/yr)	**38.5**	**25.8**	**8.6**
Phosphorus loss			
P content in topsoil (kg P/ha)	**0.5**	**0.5**	**0.5**
P loss (MMT P/yr)	**19.3**	**12.9**	**4.3**

The surface runoff loss of applied inorganic phosphate fertilizer varies significantly with a number of agronomic factors. Typical runoff rates of phosphorus in European countries range from 0.2% to 6.7%, an average of 3.5% (Hart, Quin, and Nguyen 2004; Haygarth 1997). Worldwide, the maximum rate can reach 10% under certain soil characteristics and climatic condition (Waddell and Bower 1988). Roughly, the world total phosphate fertilizer application can lead to a loss of 0.5 MMT P/yr in surface runoff.

Phosphate balance in cropland
The national phosphorus balance varies significantly from one country to another, due to differences in the use of mineral fertilizers and manure, and differences in animal husbandry practices. Broadly speaking, in developing countries soils tend to be deficient in phosphorus, while in developed countries the phosphorus content of the soils is adequate or even excessive. Taking into account applications of mineral fertilizers and manure, the balance for some West European countries is positive, particularly in the Netherlands where it exceeds 39 kg P/ha each year. For the majority of Western European countries, the phosphorus balance ranges from 8.7 to 17.5 kg P/ha annually (Johnston and Steen 2000). China, one of the largest agricultural systems in transition, also achieved a positive balance around 1980 at the national level, in parallel with increasingly application of synthetic fertilizers (Jin and Portch 2001; Wang, Halbrendt, and Johnson 1996). In 2000, the national surplus of phosphorus in Chinese soils was estimated at an average of 16 kg P/ha (Liu 2005).

To balance the phosphorus budget for the world's cropland, two natural inflows of phosphorus to croplands should be taken into account. Based on the ratio of cropland area to world total land areas, weathering and atmospheric deposition contribute 2.0 MMT P/yr as inputs to world croplands.

Table 3. Phosphorus budget for the world's
cropland in recent year (2004)

Flows	Annual fluxes (MMT P)
Inputs	22.0
Weathering	**1.6**
Atmospheric deposition	**0.4**
Synthetic fertilizers	**13.8**
Organic recycling	**6.2**
Crop residues	**2.2**
Animal wastes	**2.5**
Human wastes	**1.5**
Removals	12.7
Crops	**8.2**
Crop residues	**4.5**
Losses	19.8
Erosion	**19.3**
Runoff	**0.5**
Balance	-10.5
Input shares	
Fertilizer application	**69%**
Organic recycling	**31%**
Uptake efficiency	**64%**

The world phosphorus budget for cropland is summarized in Table 3. Although the magnitudes of recycling of animal wastes and soil erosion need further verification, the budget provides a comprehensive overview on the global phosphorus flows associated with the farming sector, which is the most intensive and complicated subsystem of the anthropogenic phosphorus cycle.

2. Environmental impacts of phosphorus uses

The phosphorus related environmental issues fall into a broad range. Some of them are caused by inappropriate use of the material, some are not. Eutrophication, being regarded as the most immediate environmental consequences of extensive phosphorus

usage in contemporary societies, has received wide attention. However, it is not the whole story, and other issues deserve to be taken into consideration. Here, our discussion is devoted to a broad socio-economic context, focusing on five phosphorus related environmental issues, i.e. mineral reserve conservation, soil erosion and degradation of soil fertility, animal waste management, sewage and detergent use, and eutrophication.

Mineral conservation

Keeping the annual mining rate of about 19.5 MMT P/yr constant, the world's known phosphate reserves could be exhausted in about 120 years. Moreover, it has been projected that the utilization trend is unlikely to decline in next 30 years. It will instead probably increase at the rate of 0.7% to 1.3% annually (FAO 2000). This strongly suggests that phosphate rock, as a finite non-renewable resource, may be exhausted in a much shorter time. It has been shown that the global average P content in raw ores dropped to 29.5% in 1996 from 32.7% in 1980, and that global reserves can sustain the current mining intensity for only another eighty years (Isherwood 2000). Some phosphorus-rich deposits around the world can be exploited much sooner. China's phosphorus reserves, for instance, constitute 26% of the world's total reserve base, second only to Morocco and the Western Sahara (USGS 2006). With a high intensive extraction activity as well as losses incurred during mining, the basic reserve of the nation's phosphorus resources, i.e. 4054 million tons with average P_2O_5 content of 17–22%, could be exhausted in 64–83 years (Liu 2005). As a result, the Ministry of Land and Resources in China counts phosphorus as one of the unsustainable resources for China's development in this century. Certainly, the larger reserve base and probably more reserves to be discovered in the future guarantee a longer lifespan of the extraction. Even so, the deposits of phosphorus in the lithosphere will inevitably be depleted before new igneous or sedimentary rocks to be formed via the biogeochemical process at the timescale of millions of years.

Soil erosion

If our estimates are reasonably accurate, one of the most important results derived from the phosphorus budget is that the world cropland has lost phosphorus at a surprising rate of 10.5 MMT P each year. This massive loss from croplands is mainly caused by wind and water erosion of topsoil. Soil erosion has been recognized as one of the most serious environmental crisis suffering the world (Brown and Wolf 1984). It is estimated that there 10 million hectares of cropland are abandoned worldwide due to lack of productivity caused by the soil erosion (Pimentel 2006). Nearly 60% of present soil erosion is induced by human activity, increasing 17 per cent since the early 1900s (Yang et al. 2003).

In contrast to the erosion loss, a huge amount of phosphorus has been mobilized in cultivated soils. Contemporary scientific knowledge cannot fully explain the complex transportation of phosphorus between plant roots, soil waters, and soils particulates. More complete understanding of these processes might suggest a possibility of controllable re-mobilization of soil phosphorus that would benefit the environment via reducing both the input of fertilizers and the loss of phosphorus from soils.

Animal wastes

Livestock husbandry, in particular large intensive feedlots, has become a major problem both for recycling of organic phosphorus and for emission of phosphorus pollutants. Worldwide, the structure of animal agriculture has changed as livestock are concentrated in fewer, but larger operations (EEA 2003a; GAO 1995; Ribaudo et al. 2003). In the United States, in spite of losing nearly a fourth of the livestock operations between 1982 and 1997, the total number of animal units has remained fairly constant at about 91 to 95 million (Kellogg et al. 2000). In China, a large number of intensive feedlots appeared in suburbs and rural areas during the last decade. According to a national investigation, the output of hogs, meat chicken (broilers) and egg chicken (layers) produced by intensive feedlots and farms accounted for 23%, 48% and 44% of the national total in 1999, respectively (SEPA 2002).

As livestock operations have become fewer, larger, and more spatially concentrated in specific areas, animal wastes have also become more concentrated in those regions. This leads to a considerable phosphorus surplus in manure, as the amount of manure nutrients relative to the assimilative capacity of land available on farms for application has grown, especially in specific high-production areas (Gerber et al. 2005; OECD 2001; Poulsen et al. 1999; Steinfeld, de Hann, and Blackburn 1998). Consequently, off-farm manure export requirements are increasing.

But because of its bulk, uneven distribution, and prohibitive cost of transport beyond a limited radius, a large proportion of manure phosphorus is now subject to disposal instead of recycling. If construction of necessary infrastructures for appropriate disposal of manure lags behind, animal wastes become a major source of phosphorus loads in surface waters. Uncontrolled phosphorus emission from intensive feedlots and farms in China has escalated in parallel with the gradual growth in total animal feeding operations and the rapid shift in breeding structure (Liu, Mol, and Chen 2004). The emission of China's livestock was estimated at 36% of its national phosphorus load to aquatic environments in 2000 (Liu 2005). Thus, livestock husbandry is the most significant source of phosphorus flux to surface waters in China, similar to the situation in European countries in the early 1990s (Morse, Lester, and Perry 1993).

Sewage treatment

In the 1960s many developed countries began to alleviate the pollution in surface water by constructing municipal sewage infrastructures and implementing phosphorus discharge restrictions on production sectors (Moss 2000; Stauffer 1998). The giant infrastructure of centralized wastewater treatment has drastically reshaped the phosphorus cycle within modern cities. Despite high economic costs, its environmental benefits in regard with removal of phosphorus from wastewater are far less than satisfactory worldwide. However, some progress has been achieved in European countries (EEA 2003b; Farmer 2001), and the United States (Litke 1999). As the centralized control strategy just removes 'pollutants' into sewage sludge rather than promotes a recovery and recycling of resources, including phosphorus, it does not really solve the long-term problem (Beck 1997; Stauffer 1998). The costly and rigid infrastructures have significantly reduced agricultural reuse of urban human excreta and contributed to a disconnect of nutrient cycles between urban areas and croplands (Foster 1999). Unfortunately, no available technologies for stable recovery and recycling of phosphorus are likely to be successfully commercialized in the near future (Driver 1998; Piekema and Roeleveld 2001; SCOPE 2001). Hence, most of the phosphorus in urban human wastes is not subject to efficient recycling and is permanently lost from the land (EEA 1997; USEPA 1999).

Proposals for recovery of phosphorus via decentralized source-separated strategies have received increasing attention since the mid 1990s (Beck 1997; Berndtsson and Hyvonen 2002; IWA and GTZ 2004; Larsen and Gujer 1996; Otterpohl, Albold, and Oldenburg 1999; Wilsenach and van Loosdrecht 2003). This decentralized and downsized sanitation concept, focused on ecologically sustainable and economically feasible closed-loop systems rather than on expensive end-of-pipe technologies, advances a new philosophy. It departs from the one-way flow of excreta from terrestrial to aquatic environments, as introduced by the conventional flush-and-discharge sewage system. The new alternative separates nutrients and domestic used water at source and handles both components individually based on material flows approaches. Thus, it avoids the disadvantages of conventional wastewater solutions and enables and facilitates nutrient recycling. Although the reinvention and transition of urban wastewater systems poses a major challenge, it does provide a promising prospect for future phosphorus recovery and recycling in an ecological and economic efficient way (Larsen et al. 2001; Pahl-Wostl et al. 2003).[2]

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[2] Detailed studies are essential as a first step, *inter alia*, of technological, organizational, economic and social aspects. In addition, the involvement of multi-stakeholders, such as residents, building owners, farmers, politicians, officials and other interested parties from the start seems essential. All these problems cannot be solved overnight, as it requires nothing less than a paradigmatic change of a large socio-technical system.

Detergents use

The use of sodium tripolyphosphate ($Na_5P_3O_{10}$, STPP), the most widely used detergent additive, has been identified as a significant contributor to eutrophication. STPP was first introduced in the US in 1946 (Emsley 2000). After reaching a peak in the 1960s, global production has finally fallen down to one half of the peak level, about 1.0 MMT P/yr, mainly due to bans on phosphorus-containing detergents in developed countries. In the late 1990s, phosphorus-free detergents accounted for 45%, 97% and 100%, respectively, in US, Japan and European countries (Litke 1999; Moss 2000).

There has been a controversy on the environmental impacts of STPP since the middle 1980s (Hoffman and Bishop 1994; Lee and Jones 1986; Liu, Mol and Chen 2004; Morse, Lester, and Perry 1994; Wilson and Jones 1994; Wilson and Jones 1995). Today, it is acknowledged that limiting or banning household consumption of phosphorus-containing detergents would not lead to a significant or a perceivable improvement of eutrophication. It would have little impact on environment and human health compared to other substitute chemicals (sodium carbonates, sodium silicates or zeolites A, and sodium nitrilotriacetate), both from an environmental and an economic perspective. In parallel with these discoveries, some Nordic countries eco-labeled STPP as an environmentally friendly component of detergents in 1997, and have re-promoted the production and consumption of STPP since then.

Eutrophication

Eutrophication is an unwanted explosion of living aquatic-based organisms in lakes and estuaries which results in oxygen depletion that can destroy an aquatic ecosystem. It has been regarded as the most important environmental problem caused by phosphorus losses. Significant eutrophication took place in the 1950s in the Great Lakes of North America and has been prevalent in many lakes and estuaries around the world (ILEC 2003; UNEP 1994). Phosphorus is often the limiting factor responsible for eutrophication, since nitrogen fluxes to waterbodies are relative large.

Phosphorus losses from industries, farmlands, animal farms and households constitute the main sources. It is helpful to discriminate phosphorus loads from different sources. Normally, point sources refer to the discharges from industry and urban wastewater. Diffuse sources (non-point sources) include background losses, losses from agriculture, losses from scattered dwellings and atmospheric deposition on water bodies. Application of area-specific indicators enables a comparison of phosphorus loads between different geographic boundaries (EEA 2005). Figure 4 illustrates a cross-county comparison of the phosphorus loads in European countries and in China to their domestic aquatic environments. Figure 5 shows a comparative histogram of the phosphorus loads to some main lakes and river basins in Europe and China.
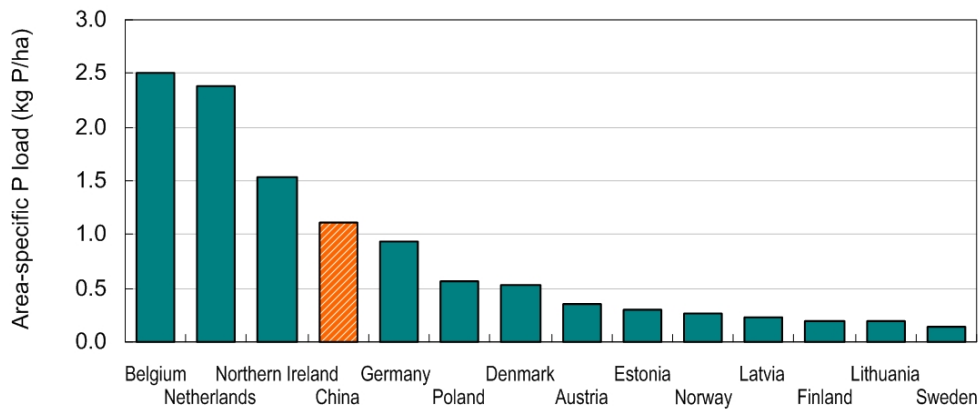
Figure 4. Comparison of phosphorus loads to aquatic environments by country (unit: kg P/ha)
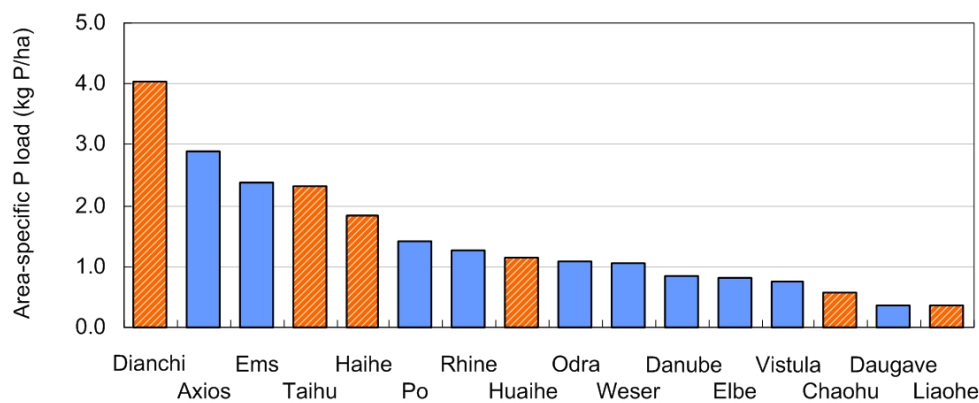
Sources: (EEA 2005; Liu 2005)

Figure 5. Comparison of phosphorus loads to aquatic environments by river basin (unit: kg P/ha)

Sources: (EEA 2005; Liu 2005; Zhang and Chen 2003)

The results show that the phosphorus loads range from 0.2 kg P/ha in Sweden to 2.5 kg P/ha in Belgium at the national level. China lies between Germany and Northern Ireland in turns of the load per unit land area. At the basin level, the phosphorus loads of the three rivers and three lakes[3] in China average 1.7 kg P/ha, 1.3 times the average of the selected European river basins and lakes. These results suggest that the Chinese economy is in general processing phosphorus 'wastes' less efficiently than developed countries. However, it is nearly impossible to determine a common benchmark (of a desired phosphorus load) to prevent waterbodies from eutrophication. This is because the complex interrelations between the amount of aquatic biomass and the phosphorus

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[3] Three rivers refer Huai River, Hai River and Liao River and three lakes include Taihu Lake, Chaohu Lake and Dianchi Lake. All of them are regarded as national key areas for water environmental protection. These six watersheds were upgraded as the national key environmental protection areas on the Forth National Environmental Protection Conference in 1996.

load are affected by a number of hydrological, meteorological and biochemical factors that remain unclear under current knowledge. Nevertheless, the comparison of area-specific phosphorus loads serves itself as a valuable analysis tool for decision making.

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