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Key Points:
- P fluxes in water-agro-food systems were studied at nested scales
- Catchment and macroregion features define contrasting P pathways/measures
- Contrasted trajectories of the P cycle over the last 50 years are studied

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Phosphorus budget in the water-agro-food system at nested scales in two contrasted regions of the world (ASEAN-8 and EU-27)

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Abstract Phosphorus (P) plays a strategic role in agricultural production as well as in the occurrence of freshwater and marine eutrophication episodes throughout the world. Moreover, the scarcity and uneven distribution of minable P resources is raising concerns about the sustainability of long-term exploitation. In this paper we analyze the P cycle in anthropic systems with an original multiscale approach (world region, country, and large basin scales) in two contrasting world regions representative of different trajectories in socioeconomic development for the 1961–2009 period: Europe (EU-27)/France and the Seine River Basin, and Asia (ASEAN-8)/Vietnam and the Red River Basin. Our approach highlights different trends in the agricultural and food production systems of the two regions. Whereas crop production increased until the 1980s in Europe and France and has stabilized thereafter, in ASEAN-8 and Vietnam it began to increase in the 1980s and it is still rising today. These trends are related to the increasing use of fertilizers, although in European countries the amount of fertilizers sharply decreased after the 1980s. On average, the total P delivered from rivers to the sea is 3 times higher for ASEAN-8 (300 kg P km^{-2} yr^{-1}) than for EU-27 countries (100 kg P km^{-2} yr^{-1}) and is twice as high in the Red River (200 kg P km^{-2} yr^{-1}) than in the Seine River (110 kg P km^{-2} yr^{-1}), with agricultural losses to water in ASEAN-8 3 times higher than in EU-27. Based on the P flux budgets, this study discusses early warnings and management options according to the particularities of the two world regions, newly integrating the perspective of surface water quality with agricultural issues (fertilizers, crop production, and surplus), food/feed exchanges, and diet, defining the so-called water-agro-food system.

1. Introduction

The lack of phosphorus (P) in soils was generally a limiting factor for European agriculture during the nineteenth century and would have been a threat in the mid-1900s in most industrial countries without the discovery of the commercial manufacturing process for phosphatic fertilizers in the 1840s [*Boulaine*, 2006; *Dawson and Hilton*, 2011]. As an irreplaceable nutrient for plants and animals, P together with nitrogen (N) is essential to sustaining agricultural production. But differently from N, for which fertilizers can be industrially produced by the Haber-Bosch process and that can be biologically fixed from the practically inexhaustible atmospheric N_2, P offers very limited local resources (besides soil stocks and landscape transfers) and has to be mined from finite and nonrenewable reserves restricted to a few countries (e.g., Morocco, China, and the United States) [*Elser and Bennett*, 2011; *Jasinski*, 2011; *Smil*, 2000].

Significant changes occurred in the global P cycle after World War II. On one hand, the remarkable increase in the population, changes in lifestyle (improved diet and use of P in detergents) and housing (increased household connection to sewage systems), and the acceleration of urbanization and industrialization have led to a huge rise in urban point release of P to surface water bodies. On the other hand, the intensification of agriculture and the extensive use of chemical fertilizers have resulted in a dramatic increase of P diffuse sources. Diffuse and point sources transferred from terrestrial to aquatic ecosystems have contributed to the development of eutrophication problems in surface waters worldwide [*Barroin*, 1980; *Carpenter et al.,* 1998; *Dorioz et al.,* 1998; *Fee*, 1979; *Garnier et al.,* 1995, 2005; *Sharpley*, 1995; *Vollenweider*, 1968].

Throughout the twentieth century and until recently, to feed the increasing population, modern agriculture in developed countries has produced a high P soil status, resulting in P accumulation in soils [*Bennett et al.,* 2001; *MacDonald et al.,* 2012a, 2012b; *Sattari et al.,* 2012]. Some studies have shown that at a critical level of P saturation in the soil surface layer, P mobility and hence leaching export can increase [*Heckrath et al.,* 1995; *Vadas and Sims*, 2014]. In Europe, intensive livestock farming systems generate the highest P surpluses (up to 200 kg ha^{-1} yr^{-1}) due to manure spreading, especially in outdoor pig farms [*Ekholm et al.,* 2005; *Lemercier et al.,* 2008; *Nielsen and Kristensen*, 2005]. Developing countries are experiencing a different situation and soils sometimes remain deficient in P [*MacDonald et al.,* 2011]. While farmers in poor countries (e.g., sub-Saharan Africa), often with P-deficient soils, cannot afford to access the fertilizer market [*Cordell et al.,* 2009; *Lougheed*, 2011], in Asian countries, the use of P mineral fertilizer has progressively increased in the past few years [*Sattari et al.,* 2012].

Despite the P abundance in most agricultural soils in developed countries and P excesses in many surface waters, the scarcity of minable P to sustain the global crop production in the coming decades remains a controversial debate [*Cordell et al.,* 2009; *Dumas et al.,* 2011; *Van Vuuren et al.,* 2010]. Given the projection of an increasing population and changing dietary habits, several studies foresee an increase in the demand and cost of P, and a shortage of minable P in the short term (50–100 years, [*Cordell et al.,* 2009; *U.S. Geological Survey*, 2010]). Other authors, however, considering future P requirements instead of current production [*Sattari et al.,* 2012] or new mining technologies and P management [*International Fertilizer Development Center,* 2010; from *Dawson and Hilton*, 2011], estimate that this shortage could occur at a longer term (up to 300–400 years).

To address the question of a more sustainable use of P resources, several scientific studies have analyzed P flows and stocks at the national, continental, or planetary scales [*Cooper and Carliell-Marquet*, 2013; *Koppelaar and Weikard*, 2013; *Sattari et al.,* 2014; *Senthilkumar et al.,* 2012; *Smil*, 2000; *Van Vuuren et al.,* 2010]. These levels of investigation are all necessary to inform policy makers and to elaborate appropriate frameworks for P management (i.e., guidelines, regulations, and policy recommendations) [*Tóth et al.,* 2014]. However, these global and national studies on P fluxes have to be integrated with local and regional investigations, closer to local needs and with an ample leeway for action.

To integrate the agricultural and food production system and the relative impacts on water resources, we attempted to consider the whole system, which we refer to as the water-agro-food system. In addition, we propose that the river basin (≈20,000 km^2 or more, i.e., medium-large according to *Harrison et al.* [2005]) is a suitable functional scale for both implementation of measures and comprehensive impact assessment studies, integrating agricultural production and the water cycle [*Billen et al.,* 2013; *Lassaletta et al.,* 2012].

First of all, the river basin is the appropriate spatial scale to understand the anthropogenic pressures and impacts on water resources. Moreover, in many countries river basins already count on administrative structures devoted to water resource management. For example, in the European Union, the Water Framework Directive [*Water Framework Directive,* 2000] has adopted the river basin as the relevant unit to apply water policy, through the development of River Basin Management Plans, and before, international commissions for the protection of transboundary rivers were created around the world at the river basin scale (e.g., for the Rhine, Danube, Mekong, Euphrates, and Tigris Rivers, respectively, in 1950, 1994, 1957, and 1946). Second, river basins can be considered as integrative socioecosystems. For historical reasons, they often correspond to the influence area of one or a few large cities that act as magnets for the watershed's socioeconomic development, structuring of regional economic activities, distribution of the population, the transport network (roads, railways, and waterways), and political power. Moreover, P sustainability requires complementarities within local agro-industrial systems to be designed [*Frosch and Gallopoulos*, 1989]. Medium-large river basins therefore appear to be an appropriate spatial unit to analyze the functioning of a comprehensive system, including urban needs, agricultural, and food production issues, and the relative impacts on water resources. River basins are also useful to envisage the implementation of specific measures to improve P sustainability.

Within this context, the objective of this paper is to investigate the water-agro-food system through the description of the P budget in two contrasting regions, chosen according to previous results found for N at the global [*Lassaletta et al.,* 2014a, 2014b], country, and river basin scales, each of them largely affected by

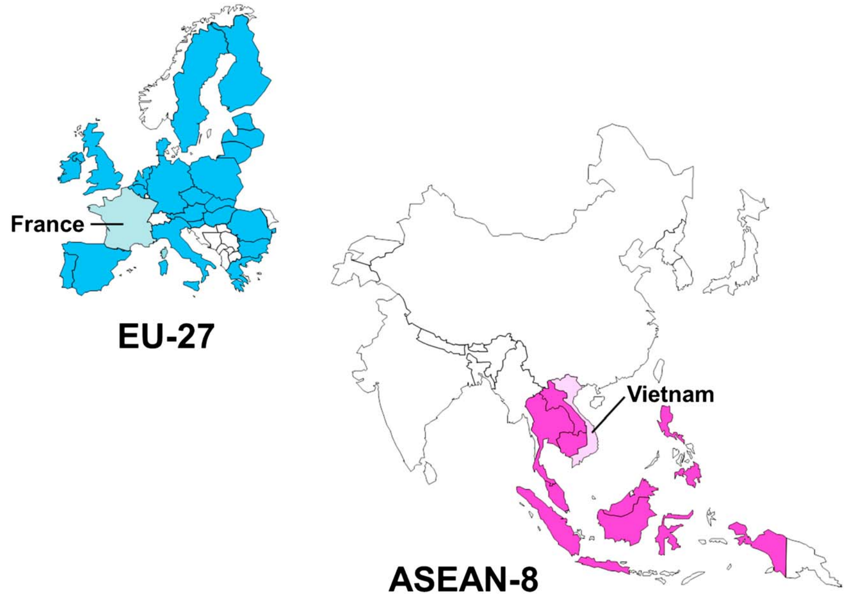

Figure 1. Map of the assembled countries considered (ASEAN-8: Indonesia, Malaysia, Philippines, Thailand, Vietnam, Lao, Cambodia, and Myanmar**;** EU-27: Austria, Belgium, Bulgaria, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, and United Kingdom.

anthropogenic pressures and representative of two different trajectories in socioeconomic development: the European Union (EU-27) and the Seine River Basin in France versus the Association of Southeast Asian Nations (ASEAN-8) and the Red River Basin in Vietnam. The EU-27 and ASEAN-8 water-agro-food systems were analyzed for the 1961–2009 period and the water-agro-food systems of the Seine and the Red River Basins for the 2000s. Analyzing the P budget at nested scales (world regions, national level, and river basin scale), this paper provides an integrative view of P for the whole water-agro-food system and notes some early warnings and management options specific to the regions and river basins.

2. Methodology for Calculating the Annual P Budget at the Selected Scales

2.1. The European and ASEAN Scales

ASEAN-8 (Indonesia, Malaysia, Philippines, Thailand, Vietnam, Lao, Cambodia, and Myanmar) and *EU-27* (Austria, Belgium, Bulgaria, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden, and United Kingdom) cover 4.5 and 4.3 Mkm^2, with 20 and 25% agricultural areas, and 540 and 487 million inhabitants, respectively (Figure 1). Gross domestic product (GDP) differs by a factor of about 10 (Figure 2).

We calculated an overall P budget for the 27 countries of Europe and for the 8 ASEAN countries, from 1961 to 2009, in order to analyze the changes during this period. The GlobalNEWS data [*Seitzinger et al.,* 2005, 2010] were used to calculate water budgets, whereas FAOSTAT data (http://faostat.fao.org/) were processed and analyzed for the construction of the agro-food budget. Until 1991, Estonia, Latvia, and Lithuania were part of the Soviet Union, and Slovenia was part of Yugoslavia. Before that date, the available information (FAO data) is not disaggregated; therefore, they were not taken into account in the results previous to 1992. These countries account for 5% of the EU-27 surface and 2.4% of its current population.

We documented the total anthropogenic P inputs entering the countries, i.e., synthetic fertilizers, net international import of food and feed, and use of detergents, which were summarized to obtain an assembled figure for each of the two world regions. The results were expressed in P input per square kilometer of their regional surface area in order to compare the different fluxes (water, soils, feed and food, etc.).

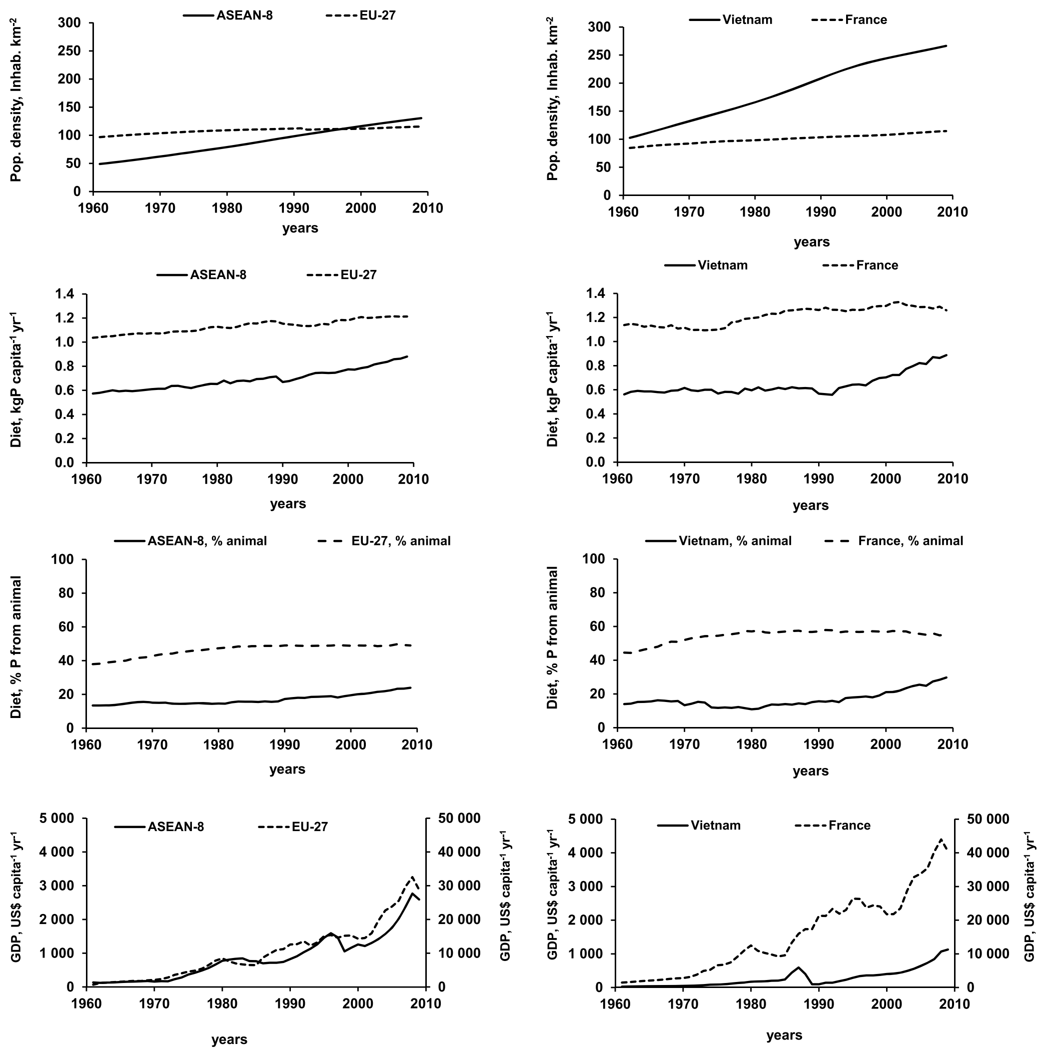

Figure 2. Changes in population density and P human consumption (left column) in ASEAN-8 and EU-27 and (right column) in Vietnam and France from 1961 to 2009 (FAOSTAT). Change in GDP during the same period.

2.1.1. P Budget of the Agro-Food System

Estimations of atmospheric P (1) input were based on the values provided by *Vet et al.* [2014], partly based on *Mahowald et al.* [2008], and represent 10 kg P km^{-2} yr^{-1} and 5 kg P km^{-2} yr^{-1} for EU-27 and ASEAN-8, respectively, including wet and dry deposition.

Yearly data on *inorganic P applied as crop fertilizer (2)* for the entire five decade period were obtained from the Resources module of the FAOSTAT database. Occasional gaps were completed with data from the International Fertilizer Industry Association (http://www.fertilizer.org/).

We also estimated the total yearly agricultural P uptake and export in harvested products of the 176 primary crops cultivated in all countries. The P content was assigned to each product, and this was multiplied by their total production as given in the Production Module of the FAOSTAT database. The crop production of each country was added to obtain the region's total primary *agricultural production (3)*.

Total *animal P production (4)* was estimated using the 18 categories of meat, 5 categories of milk, and 1 category of eggs provided in the "Livestock Primary" module of the FAOSTAT database, and multiplying each category by its corresponding P content.

P excreted by livestock (5) was estimated using the excretion factors provided by *Van der Hoek* [1998] at the regional level for the different types of animal provided in the FAOSTAT in terms of number of heads.

These factors expressed as N were transformed to P according to the N:P ratios for animal excreta used by *Hong et al.* [2012]. Dairy and nondairy cows were separated using the "Livestock processed" module (FAOSTAT). The proportion of P excreted that is finally used as manure applied to cropland was taken from the estimates of *Sheldrick et al.* [2003] at the regional level. The efficiency of vegetal into animal P transformation was estimated by dividing the animal production by the total animal excretion plus the animal production.

The net import of P embedded in internationally traded agricultural commodities (6) was calculated from data provided by the Trade Module of the FAOSTAT database (http://faostat.fao.org/). The module contains information on commodities traded across world countries, including food, feed, and fiber (see detailed methodology in *Lassaletta et al.* [2014a]). P content of major products was gathered (255 vegetal products for food and feed, 99 animal products, and 30 products used as materials) from several sources [*Soltner*, 1979; *Le et al.,* 2005; http://dietgrail.com/phosphorus/; http://plants.usda.gov/java/; http://wholefoodcatalog.info/foods/]. When estimating the total net import/export in the two regions (EU-27 and ASEAN-8), the internal flows between countries belonging to the same region are automatically discounted from the total.

P in the human diet (7), documented as vegetal and animal consumption, was estimated using the Food Supply FAOstat module, which was transformed into P contents. Human excretion was assumed equal to ingestion [*Jönsson et al.,* 2004].

To calculate *the detergent consumption (8)*, for EU-27, we considered a value of 0.224 kg P cap^{-1} yr^{-1}, which corresponds to the figure for EU-18 for the 2000s and 0.373 kg P cap^{-1} yr^{-1} for France [*Wind*, 2007]. For ASEAN-8, taking into account the detergent consumption (4 kg cap^{-1} yr^{-1} for Thailand and Malaysia, 2 kg cap^{-1} yr^{-1} for Indonesia) [*Hochiminh University of Industry (HUI)*, 2010] and considering 3–5% P kg^{-1} detergent (cf. *Wind* [2007] for EU-18), we applied, respectively, the highest and lowest consumptions to the countries with, respectively, the highest and lowest GDPs, and finally a value of 0.09–0.15 kg P cap^{-1} yr^{-1} for ASEAN-8 and 0.06–0.1 kg P cap^{-1} yr^{-1} for Vietnam.

P from domestic wastewater returning to soils (9) was calculated as the sum of human excretion (7) + detergent (8) minus the amount emitted to the river (cf (10) below). For ASEAN-8, we considered that the P fraction of 3.8% (GlobalNEWS data) for human excretion and detergent removed by wastewater treatment plants (WWTPs) all returned to soils. For EU-27 we considered that 42% of P from the domestic wastewaters are recycled to soils [*Science Communication Unit (SCU),* 2013].

2.1.2. P Budget of the Hydrosystem

To compute the P fluxes of the water system, we used the results of the GlobalNEWS model [*Seitzinger et al.,* 2010]. Among the 6292 watersheds of the GlobalNEWS database, 590 and 467 were selected in EU-27 and ASEAN-8, respectively. The difference in the surface areas of land resulting from the sum of the watersheds (GlobalNEWS) and those obtained from the sum of the surface areas of the countries (FAO) were 1% for EU-27 and 7% for ASEAN-8.

The GlobalNEWS model outcomes contain the amount of *P from human excretion and detergent emitted to rivers (10)* for each region (29.9 and 27.1 kg P km^{-2} yr^{-1} for EU-27 and ASEAN-8, respectively).

At the watershed scale, the GlobalNEWS model also provides the fluxes of dissolved inorganic phosphorus, dissolved organic phosphorus, and particulate phosphorus (PP) (in kg P km^{-2} yr^{-1}) *delivered at the river outlets (11)*. These fluxes were summed for all the watersheds contained in the two regions of the world selected.

P fluxes resulting from erosion of nonagricultural (12) and *agricultural soils (13)* were assessed using the erosion rates proposed by *Cerdan et al.* [2010] for EU-27 and *Valentin et al.* [2008] for ASEAN-8, taking into account the proportion of land uses (in terms of agricultural soils, forest, and grassland) and the associated P content measured in a variety of soils in the Seine and the Red River basins, respectively. As shown by *Némery and Garnier* [2007a], we assumed that the P content in suspended solids of river water is close to the P content in soils, especially with increasing discharge. Rock weathering is considered as a process making new P available in the soil [*Ruttenberg*, 2014]. Assuming a steady state for nonagricultural (seminatural) soils, we roughly estimated the *rate of P weathering (14)* as equal to their erosion rate. We used the rate per unit area as an input by *rock weathering to agricultural soil (15)*. This is likely to be an overestimation because cultivated vegetation is less active than natural vegetation in promoting rock weathering. *P retention (16)* within the watershed was calculated as the difference between the inputs from nonagricultural and agricultural soils, plus those from treated and untreated wastewater, and the deliveries at the river mouth. The *P accumulation in soils (17)* was here defined as the difference between P inputs, including rock weathering, fertilizer, and animal manure/sludge/biosolid application, and export, including harvesting, grazing by domestic animals, and erosion.

Also, P agricultural surpluses were calculated as the difference between total P inputs to croplands (rock weathering, inorganic fertilizers + manure) minus the total crop output. The cropland area of each region is defined as the sum of the surface areas of all individual crops (expressed in hectares of agricultural land), except when this sum exceeds the total "arable land and permanent crop surface" provided by the FAOSTAT resource module, in which case we used the value of this FAOSTAT variable.

2.2. The River Basin Scale: The Seine River (France) and the Red River (Vietnam)

Besides considering two large regions of the world, we focused on the Seine River in France and the Red River in Vietnam, for which we were able to provide the P budget of the water-agro-food system, based on previously published local studies [*Le et al.,* 2005; *Luu et al.,* 2012; *Némery and Garnier*, 2007a, 2007b]. Unlike our previous publications, here we did not consider the upstream basins and the estuary/delta separately but merged them.

The Seine River Basin, in the north of France, has 17.5 million inhabitants, of which 11.8 million live in the Paris agglomeration and 1 million in the estuarine area, 200 km downstream from Paris. The surface area of the Seine basin is 76,200 km^2, including the Seine estuary watershed covering 4610 km^2. The estuarine zone accounts for about 6% of the total surface area of the basin and 6% of the population. Land use is dominated by intensively cropped arable land (52%). Grassland areas account for 11% and forest 25%. The Seine basin covers 12% of the French territory (675,417 km^2) but is home to 27% of its population.

The population in the Red River, in the north of Vietnam, reaches approximately 35 million inhabitants including 16.6 million living in the delta. The Red River basin has a surface area of about 161,000 km^2 and 14,300 km^2 for the delta, i.e., 8.4% of the basin area, but where 35% of the population lives. Agricultural land occupies 33% of the upstream basin and 47% of the delta (including aquaculture). Forests, which dominate in the upstream basin (54%), are reduced to 13% in the delta where urban infrastructures occupy 21% of the land. With 12% of surface water in the delta and 5% of rocky areas in the upstream basin, a small part is left to grassland. The Vietnamese part of the Red River Basin accounts for about 25% of the country (331,698 km^2) and includes up to 40% of the population.

The GDP of France and Vietnam differ by a factor of 10, similar to the values obtained at the regional level between EU-27 and ASEAN-8.

We inventoried the sources and sinks of P at the basin scale, in the soil and aquatic compartments of the Seine and the Red Rivers, including the estuary and the delta.

2.2.1. P Budget of the Agro-Food System

Data such as inputs to land *(atmospheric deposition (1)*, *fertilizers (2)*, *agricultural production (3)*, *livestock production (4)*, *excretion (5)*, and *net import (6)* available at the district level (in Vietnam) and the canton level (in France) were gathered for each basin and then calculated *pro rata* considering the surface area located within each basin. All these fluxes are reported in Table 2 (see also *Le et al.* [2005] and *Luu et al.* [2012] for the Red River and *Némery and Garnier* [2007a, 2007b] for the Seine River).

However, some figures were newly gathered or revised according to recent literature.

For *atmospheric deposition (1)*, in the Red River Basin we selected the value used for ASEAN-8 of 5 kg P km^{-2} yr^{-1} (rather than the value of 60 kg P km^{-2} yr^{-1} provided by *Le et al.* [2005] on the basis of data from China, Africa, and Ecuador). In the Seine Basin we used the value of 35 kg P km^{-2} yr^{-1} [*Némery and Garnier*, 2007a].

P in the human diet (7) for the Seine and Red River Basins are those calculated for France and Vietnam using the Food Supply FAOstat module, human excretion being assumed equal to ingestion (see above).

Detergent consumption (8) was newly calculated in the Seine Basin according to the per capita value of 0.373 kg P yr^{-1} for France and the corresponding population in the basin [*Wind*, 2007], whereas for the Red River Basin we used 0.06–0.1 kg P cap^{-1}/yr^{-1} for Vietnam according to *HUI* [2010] (see above).

For P in wastewater recycled to soils (9), the calculations differed between the Seine and the Red River. In the Seine Basin, we used French Environmental Statistics on the fate of WWTP sludge provided in dry weight per department (http://www.stats.environnement.developpement-durable.gouv.fr/Eider/series.do). Taking into account 50.7% of dry matter in the raw sludge and 5.2% of P_2O_5 in the raw material, we estimated that 150 kg P km^{-2} yr^{-1} were produced. *Institut d'Aménagement et d'Urbanisme de la Région d'Ile-de-France (IAURIF)* [2003] mentioned that 30% of the sludge produced in the Paris conurbation, versus 60% in France, was used in agriculture so that we calculated a weighted average of ≈ 40% for the Seine Basin (i.e., considering Paris agglomeration and the fraction of the 28 administrative departments in the basin), a figure that closely agrees with the 42% provided for EU-27 [*SCU,* 2013]. To balance the budget, we had to take into account that wastewater pretreatment eliminates a part of P not considered as sludge (i.e., 140 kg P km^{-2} yr^{-1}). For the Red River, without sanitation facilities for the period considered, we assumed a rate of P wastewater recycled as in *Le et al.* [2005] for the upstream basin (75% from the 18 million equivalent (equ.) inhabitants) and as in *Luu et al.* [2012] for the delta (25% from the 16.6 million equ. inhabitants).

By the difference between the total P in human sewage (excretion + detergent) and the P amount discharged into the drainage network or reused in agriculture, we estimated the P removed in WWTPs.

2.2.2. P Budget of the Hydrosystem

In the Seine River, *P point sources discharged to the river network (10)* were estimated from an inventory of WWTPs, including their capacity and type of treatments [*Passy et al.,* 2013]. In the Red River, the assumptions for *effluent discharge to surface water (10)* are those made for P recycling to soils, i.e., 25% from the 18 million equ. inhabitants in the upstream basin versus 75% from the 16.6 million equ. inhabitants in the delta are discharged untreated (*Le et al.* [2005] and *Luu et al.* [2012], respectively). The amount of water discharged from industries and its P content were estimated from a census of the major establishments, and accounted for 27 kg P km^{-2} yr^{-1} [*Le et al.,* 2005; *Luu et al.,* 2012], to which 2 kg P km^{-2} yr^{-1} from detergents must be added, a negligible value in the 2000s in Vietnam, as mentioned in *Van Drecht et al.* [2009].

P deliveries at the river outlet (11) were calculated for both basins using P concentration measurements and discharge values provided by water agency authorities [*Némery and Garnier*, 2007a; *Le et al.,* 2005; *Luu et al.,* 2012].

For P losses from *nonagricultural soils (12)*, the values are those given in *Némery and Garnier* [2007a], *Le et al.* [2005], and *Luu et al.* [2012].

For *P lost from soils (13)*, the same approach was applied for the Seine and Europe and for the Red River and ASEAN, taking into account (i) the erosion rates proposed by *Cerdan et al.* [2010] for EU-27 and *Valentin et al.* [2008] for ASEAN-8, (ii) the proportion of land use (in terms of agricultural soils, forest, and grassland), and (iii) the associated P content measured in a variety of soils in the Seine and the Red River Basins (e.g., for cash crops: 1.25 g P kg_{soil}^{-1} in the Red River Basin and 0.9 g P kg_{soil}^{-1} in the Seine Basin). As explained above, we estimated *rock weathering under natural soils (14)* as equal to their erosion losses and assumed the *same rate under agricultural soils (15)*.

Again, *P retention (16)* was calculated as the difference between the deliveries at the outlet of the rivers and the sum of the inputs from nonagricultural and agricultural soils and from WWTPs. The *P accumulation in soils (17)* was again defined as the difference between P inputs, including rock weathering, fertilizer, and animal manure/sludge/biosolid application, and export, including harvesting, grazing by domestic animals, and erosion.

3. Results

3.1. Agro-Food Trajectories in ASEAN-8 and Vietnam Versus EU-27 and France for 1961–2009

3.1.1. Socioeconomic Trends

ASEAN-8 and EU-27 have a similar surface area, total population (Table 1), and population density (average for 2000–2005) (120 and 113 inhabitants km^{-2}, respectively). However, whereas the population has not changed much in EU-27 since the 1960s, in the same period it has increased 2.6-fold in the ASEAN-8 countries (Figure 2). France is representative of the average EU-27, but population density in Vietnam is twice the ASEAN-8 average and has increased faster in the last few years as compared to the beginning of the 1960s (Figure 2).

The difference in human diet P, almost twice as high in EU-27 as in ASEAN-8 in the 1960s, has been reduced in the last decade due to a stabilization in Europe and an increase of P consumption in Asia (0.8 kg P $capita^{-1}$ yr^{-1} in ASEAN-8 and 1.2 kg P $capita^{-1}$ yr^{-1} in EU-27 for the 2000–2005 period)

Table 1. P Agro-Food Budget for ASEAN-8 and EU-27 and for Vietnam and France[a]

		2000–2005			
	Units	ASEAN-8	EU-27	Vietnam	France
Region surface	km^2	4,507,419	4,328,777	331,051	549,190
Agricultural surface	km^2	899,679	1,088,554	88,755	165,654
Population	million inhabitants	542	488	83	60
Population density	inhabitants km^{-2}	120	113	251	110
Crop production	$kg P km^{-2} yr^{-1}$	182	365	398	624
Animal Ingestion	$kg P km^{-2} yr^{-1}$	210	525	530	641
Animal excretion	$kg P km^{-2} yr^{-1}$	203	470	512	572
Animal production	$kg P km^{-2} yr^{-1}$	7	55	17	69
Manure applied to crops	$kg P km^{-2} yr^{-1}$	85	290	215	361
Fertilizer chemicals	$kg P km^{-2} yr^{-1}$	156	353	729	573
Total Input to crops	$kg P km^{-2} yr^{-1}$	241	643	944	934
Net IMP-EXP, animal	$kg P km^{-2} yr^{-1}$	0	−2	0	−7
Net IMP-EXP, vegetal	$kg P km^{-2} yr^{-1}$	8	53	−15	−135
Net IMP-EXP, fiber	$kg P km^{-2} yr^{-1}$	1	1	1	0
Human diet	$kg P km^{-2} yr^{-1}$	96	135	190	143
Animal P	$kg P km^{-2} yr^{-1}$	20	66	44	81
Vegetal P	$kg P km^{-2} yr^{-1}$	76	69	146	62

[a]Data sources: FAOSTAT.

(Figure 2). The proportion of P from animal products in the diet of ASEAN people (20% animal) is still 2.5-fold lower than that of Europeans (50% animal), although this proportion has been increasing in ASEAN-8. Generally, the trends for France and Vietnam are similar to those of their respective regions (Figure 2).

Although a similar GDP trajectory has been observed for ASEAN-8 and EU-27 since the 1960s, with an increase by a factor of 20 and 40, respectively, GDP in Europe was on average 12 times higher than in ASEAN countries during the 1961–2009 period (Figure 2). Despite an overall increase in GDP during the same period for France and Vietnam, the differences between these two countries have amplified, France being among the richest countries in EU-27 and Vietnam one of the poorest in ASEAN-8 (Figure 2).

3.1.2. Agricultural Trends

Total crop production in terms of P progressively increased by a factor of 5 in the ASEAN-8 countries from 1961 to 2009 and by a factor of 2 in EU-27, with crop production stabilizing since the middle of the 1980s; however, crop production in EU-27 remains about 1.6 times that in ASEAN-8 (Figure 3). Interestingly, P application on crops (including chemical fertilizers and manure) has increased by the same factor of 5 in ASEAN-8, but it has been reduced by half in EU-27 in the last few years (compared to the maximum fertilization of the 1970–1990 period) without any reduction in crop production, even in recent years when a further reduction in fertilizers has occurred (Figure 3). In ASEAN-8 the agricultural surface has almost doubled in the period studied, reaching the same extent (10×10^6 ha) as in EU-27, where instead it remained rather stable in time. At the country level, P input trends for both France and Vietnam are similar to those in their respective regions, but P application is almost twice their respective regional average values, with a sharp increase in fertilization in Vietnam since 1990. Total fertilization includes mineral fertilizers and manure, and it is worth noting that the fraction of manure is increasing in EU-27 (and France), while it is decreasing in ASEAN-8 (and Vietnam) (Figure 3).

As a result of these trends, European and French P surpluses in agricultural soils were maximum during the peak of fertilization in 1970–1990 and have drastically decreased since then (from 40 to <10 kg P ha^{-1} yr^{-1} in EU-27 and from 60 to <10 kg P ha^{-1} yr^{-1} in France, even close to zero in 2008 and 2009, Figure 3). In ASEAN-8 countries, the surplus reached a maximum of 5 kg P ha^{-1} yr^{-1} during the 1990–2000 decade and it is currently close to zero, whereas in Vietnam the surplus started to increase some time later and remains at about 20 kg P ha^{-1} yr^{-1} (Figure 3).

3.2. Water-Agro-Food Phosphorus Budget in ASEAN-8 and Europe-27 for 2000–2005

The P budget of the agro-food system was combined with the P budget of the hydrological system to represent all the interconnections of the water-agro-food system. Figure 4 presents the P budget of the water-agro-food system estimated for ASEAN-8 and EU-27, describing the major P fluxes.

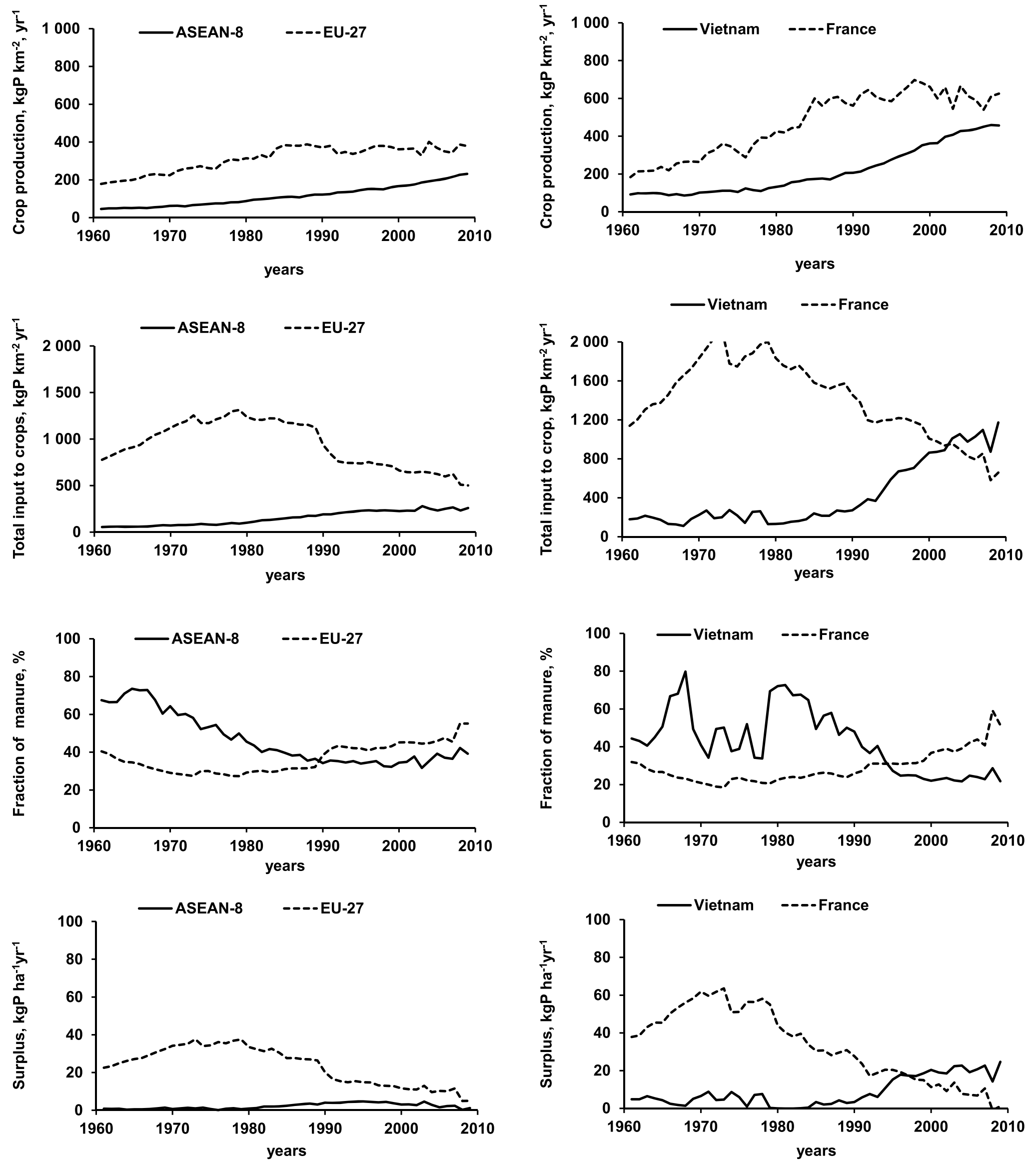

Figure 3. Changes in the crop production, total inputs to crops and its fraction of manure (left column) in ASEAN-8 and EU-27 (and (right column) in Vietnam and France from 1961 to 2009 (FAOSTAT). Surpluses (inputs − outputs of agricultural soils) are given in comparison.

3.2.1. The Agro-Food Systems

For the 2000–2005 period, mineral fertilizer used in EU-27 was more than twice that in ASEAN-8 (353 and 156 kg P km^{-2} yr^{-1}, respectively), and the same ratio was found for crop and vegetal agricultural production (crop production plus meadows and pastures) (Table 1 and Figure 4). Atmospheric P deposition is rather low compared to fertilizer input and represents in both cases about 3% of the total input to soils.

According to the net import-export of vegetal and animal products, EU-27 exports some animal products (2 kg P km^{-2} yr^{-1}) but imports vegetal crops (54 kg P km^{-2} yr^{-1}) to fulfill animal and human consumption needs. The EU-27 human diet comprises about 50% of animal P. Although human P consumption of animal products is 3 times lower in ASEAN-8 than in EU-27, ASEAN-8 has to import some animal products (0.2 kg P km^{-2} yr^{-1}) and about 10% of the vegetal products for human consumption (9 kg P km^{-2} yr^{-1}). Fish consumption, twice as high in ASEAN than in EU-27, accounts for 20% of the P animal diet for the former and only 3% for the latter (Table 1 and Figure 4).

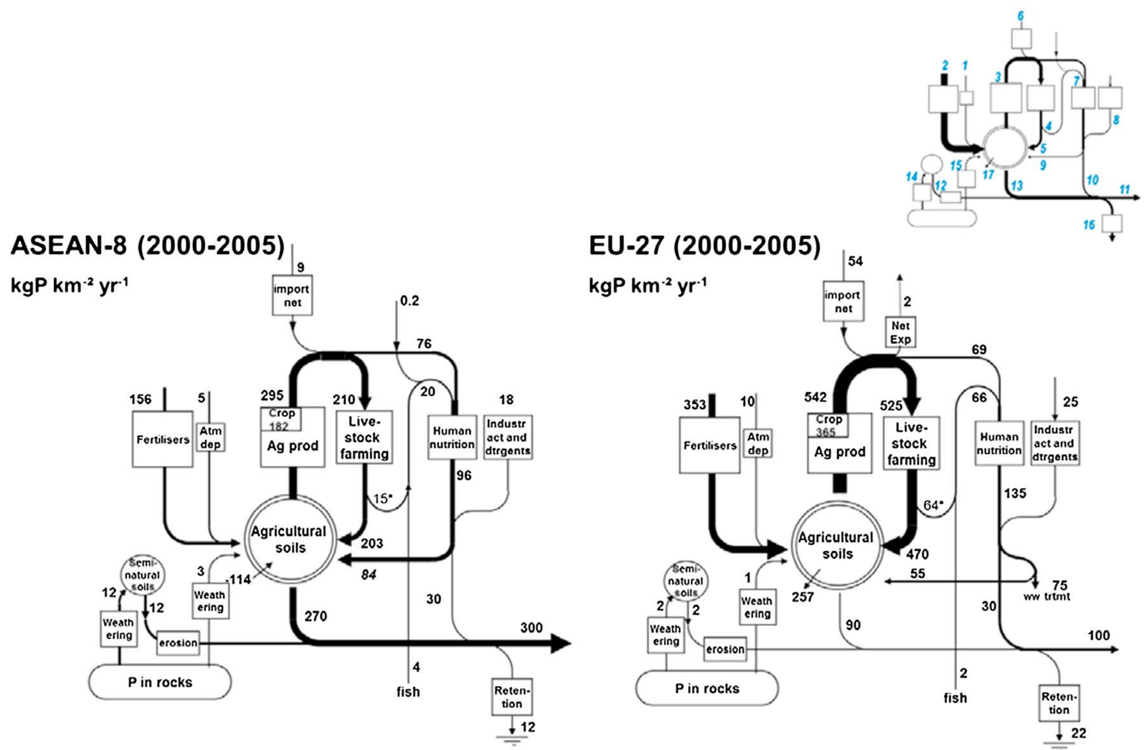

Figure 4. Water-agro-food P budget for ASEAN-8 and EU-27. The values for the agro-food systems are averaged for 2000–2005, those for the hydrosystems are from 2000. A stamp indicates the numbering of the P fluxes as cited in the text. Figures correspond to those in Table 1, rounded for equilibrating the budget, except those of animal production (asterisks) issued from FAOSTAT that were calculated from basin-scale statistical data (see text).

The statistical FAO data on animal production used for the ASEAN-8 and EU-27 budget resulted in very low estimates of the vegetal to animal P conversion (3 and 10%, respectively, Table 1). Taking into account instead the estimated human animal products consumption, we obtained vegetal to animal P conversions of 8 and 12%, values closer to the more accurate basin-scale statistical data (Figure 4 and Table 2).

3.2.2. The Hydrosystems

As computed from the GlobalNEWS data [*Seitzinger et al.,* 2010], P exported by the river network is three times higher in ASEAN-8 than EU-27, with a respective contribution of 80% and 50% of particulate P. Regarding domestic input to the drainage network, wastewater P (including human excreta, detergent, and industrial waste) discharged into surface waters is similar in the EU-27 and ASEAN-8 regions (about 30 kg P km^{-2} yr^{-1}). This results from a urban population of 58.6% and 65.9% in EU-27 and ASEAN-8, with a corresponding connection to sewage systems of 64.4% and 34.8% and P removal in sewage treatments totaling 63.4% in Europe and only 3.8% in ASEAN (Global-NEWS data). In EU-27, the largest proportion of P is retained in wastewater treatment plants (130 kg P km^{-2} yr^{-1}) and 42% of it is reused (i.e., 55 kg P km^{-2} yr^{-1}) on agricultural soils [*SCU,* 2013]. Instead, in ASEAN-8 countries, in the absence of sanitation, 75% of domestically used P returns to soils.

For ASEAN-8, P delivery at river outlets is 300 kg P km^{-2} yr^{-1}. The P lost from nonagricultural soils (e.g., wetlands and forests) and from agricultural soils amounts to 12 kg P km^{-2} yr^{-1} and 270 kg P km^{-2} yr^{-1}, respectively, to which 30 kg P km^{-2} yr^{-1} from domestic inputs have to be added to estimate total inputs (312 kg P km^{-2} yr^{-1}) to the drainage network. In EU-27, P river deliveries are much lower, i.e., 100 kg P km^{-2} yr^{-1}, with a total input of 122 kg P km^{-2} yr^{-1} (2, 90, and 30 kg P km^{-2} yr^{-1} for nonagricultural soils, agricultural soils, and domestic inputs, respectively) (Figure 4).

The P retention computed by the difference between P inputs to the drainage network and the deliveries at coastal zones is rather low for ASEAN-8 (12 kg P km^{-2} yr^{-1}, i.e., 4%) and higher for Europe (22 kg P km^{-2} yr^{-1}, i.e., 18%).

Table 2. P Budget in the Red River and Its Delta and the Seine River and Its Estuary (for the Year 2000)[a]

		2000	
Basin Description	P Fluxes	Seine Basin	Red River Basin
Area (km^2)		76,215	161,325
Agricultural area (%)		58	34
Population (millions)		18	31
Population density (inhabitants km^{-2})		230	190
Agrosystem	Atmospheric deposition (kg P km^{-2} yr^{-1})	35	*5*
	Fertilizer application (kg P km^{-2} yr^{-1})	1,229	624
	Cattle farming (kg P km^{-2} yr^{-1})		
	Meat and dairy production	67	27
	Excretion	354	235
	Grazing and feed consumption	421	283
	Agriculture and food balance (kg P km^{-2} yr^{-1})		
	Net commercial export/import	*−905*	*50*
	Net agriculture production	1,459	370
Food system	Human consumption (kg P km^{-2} yr^{-1})		
	animal	164	59
	vegetal	135	137
Hydrosystem	Inputs to the hydrosystem (kg P km^{-2} yr^{-1})		
	Domestic and industrial wastewater	*90*	*80*
	Leaching from forest soils	*2*	*13*
	Leaching and erosion from agriculture	*160*	*330*
	Total P riverine delivery (kg P km^{-2} yr^{-1})	106	195
	Retention (kg P km^{-2} yr^{-1})	146	228
	Retention/input (%)	*58*	*54*

[a]Data sources: *Le et al.* [2005], *Luu et al.* [2012], and *Némery and Garnier* [2007a, 2007b]; new estimations in italics (see text).

3.3. Water-Agro-Food Phosphorus Budget in the Seine and the Red River Basins

3.3.1. The Agro-Food Systems

The atmospheric P deposition for the Seine River is that measured by *Némery and Garnier* [2007a], higher than the deposition value found in the literature and taken for EU-27 [*Vet et al.,* 2014], whereas identical values are taken for ASEAN-8 and the Red River (Figures 4 and 5).

In the Seine and the Red River Basins, P mineral fertilizer inputs (kg P km^{-2} yr^{-1}) are 3.5 and 4 times greater than those of their respective world regions (Figures 4 and 5), but the corresponding ratios for agricultural productivity are only 1.25 and 2.25, respectively. Compared to their respective countries, the Seine Basin, typically specialized in cash crop farming, uses 1.3 times the amount of mineral fertilizers applied in France and exports 60% of its production, whereas the Red River applies approximately 35% less fertilizer than Vietnam, and imports feed and food (Figures 4 and 5). Vietnam is indeed a net exporter of agricultural products, unlike the Red River Basin, although at a rate 8 times lower than France (Tables 1 and 2).

In the Seine Basin, livestock farming is restricted to the eastern and western periphery of the basin so that to feed its 16 million inhabitants the Seine Basin has to import meat and milk, while France is globally a net exporter of animal products (Tables 1 and 2 and Figure 5). The Red River Basin also imports meat, contrary to the whole country, which has a zero net import-export balance (Table 1 and Figure 5).

3.3.2. The Hydrosystems

Regarding domestic wastewater, the pattern for the Seine and Red River basins is similar to the observations made at the EU-27 and ASEAN-8 scales. In the Seine basin, domestic P is mostly retained in WWTPs so that wastewater reaching the river network accounts for about 20% of the urban emissions. Taking into account the amount of domestic wastewater (physiological release + detergents = 380 kg P km^{-2} yr^{-1}) and the fraction discharged into the river (90 kg P km^{-2} yr^{-1}), 290 kg P km^{-2} yr^{-1} are retained in WWTPs, among which 150 kg P km^{-2} yr^{-1} as secondary sludge. About 40% of this sludge is recycled on agricultural soils (60 kg P km^{-2} yr^{-1}) leading to 230 kg P km^{-2} yr^{-1} disposed in landfills (Figure 5).

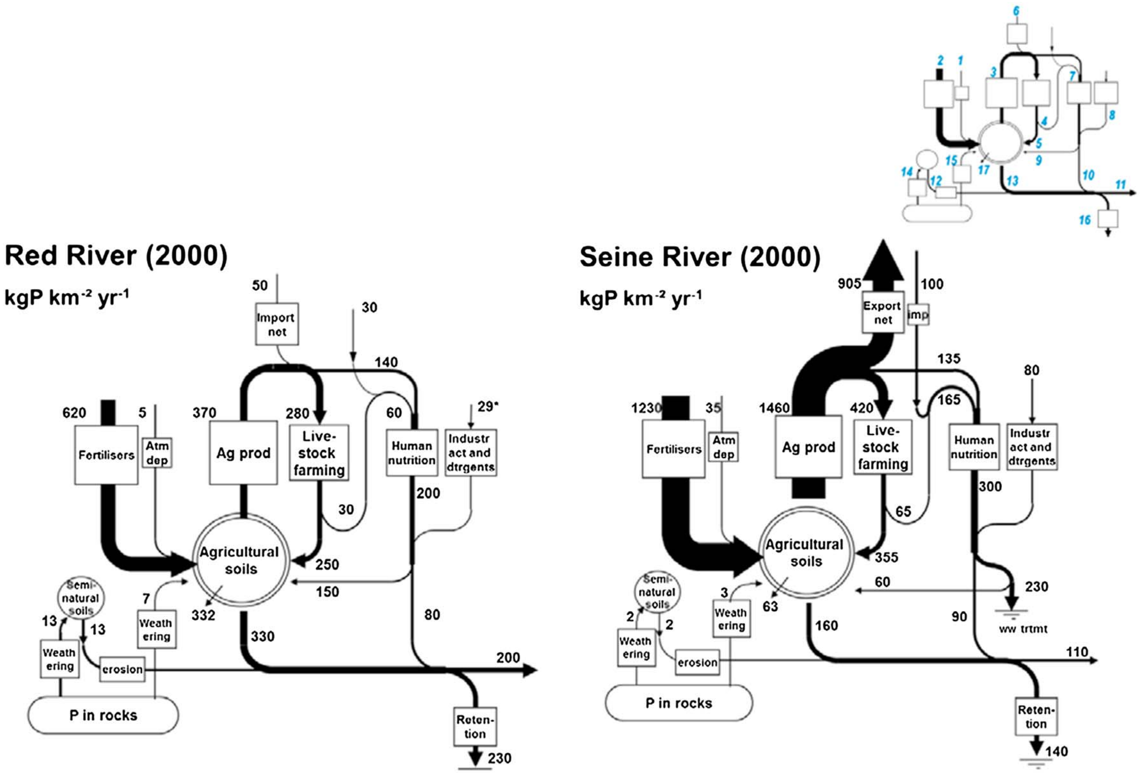

Figure 5. Water-agro-food P budget for the Seine and Red River Basins. The values for the water-agro-food systems are from 2000. A stamp indicates the numbering of the P fluxes as cited in the text. Figures correspond to those in Table 2, rounded for equilibrating the budget.

In the Red River, our estimates of detergent consumption led to the low value of 2 kg P km^{-2} yr^{-1}, to which 27 kg P km^{-2} yr^{-1} from industrial wastes must be added in addition to the 200 kg P km^{-2} yr^{-1} from physiological release to account for the potential input to the surface water. With an estimated 75% untreated domestic wastewaters discharged directly into the river in the delta area, and 25% in the upstream basin, 80 kg P km^{-2} yr^{-1} is directly discharged into surface waters and 150 kg P km^{-2} yr^{-1} is assumed to be directly reused on agricultural lands (Figure 5).

P losses from Red River soils, with a mountainous relief and monsoon climate, are about twice those of the lowland temperate Seine Basin (330 versus 160 kg P km^{-2} yr^{-1}), with a twofold higher retention in the Red River, especially in the delta [*Luu et al.,* 2012]. On the contrary, P retention occurs more in the upstream Seine Basin than in the macrotidal estuary [*Garnier et al.,* 2014; *Némery and Garnier*, 2007b]. In both cases, the rather high retention reduces by half the rate of P deliveries to the coast. Surprisingly, such high P retention values of P in the river network are not apparent from the budget calculated at regional scale (see Figure 4).

4. Discussion

The comparative P budgets presented in this study allow an analysis of the environmental issues related to P in two different socioeconomic contexts providing an integrative view of the agri-food systems and the hydrosystems. Despite the uncertainties associated with the various figures of the P budgets, their coherence indicates the robustness of the approach considering the variety of data sources used.

We will first discuss the changes observed over the last 50 years, then the associated environmental threats they are currently causing. Finally, we will propose options for improved management of P in agricultural and aquatic systems in the two regions.

4.1. Phosphorus Budget at the Regional and Watershed Scales (ASEAN-8 and EU-27 1961–2009; Red River and Seine River 2000s)

4.1.1. Fertilizer Use

The Green Revolution after the Second World War led to a tremendous increase in the use of chemical fertilizers, both N [*Billen et al.,* 2014] and P [*Cordell et al.,* 2009; *Smil*, 2000]. This stimulated a large increase in agricultural production, even though starvation problems were not eradicated [*Bacon et al.,* 2014]. P inputs to agricultural lands indeed increased by a factor of 1.5–5 during the 1961–2000 period, with a trajectory that differs in EU-27 and ASEAN-8. In EU-27, the maximum application was found in the 1980s and has decreased since then, whereas P fertilizer application is still increasing or just leveling off in all the ASEAN-8 developing countries. Accordingly, regarding the type of P fertilizers, the recent trend in EU-27 countries is an increase in the proportion of animal manure versus mineral P since the 1990s, reaching 50% of the total fertilizer inputs, as also discussed by *Senthilkumar et al.* [2012], while the opposite trend is observed in ASEAN-8 and Vietnam, with a tendency toward using proportionally more mineral commercial fertilizers. This represents an ongoing change from a traditional to a commercial production mode, as observed in Vietnam by *Montangero et al.* [2007].

4.1.2. Crop Production, Diet, and Net Input of Food and Feed

The increase in crop production supported by increasing application of N and P fertilizers has provided more food for the increasing populations and also diet changes in developing countries. For ASEAN-8, the per capita P consumption in food has increased by 34% since 1960, while only by 6% in EU-27, although the per capita consumption in ASEAN-8 still remains far below European standards. Such changes in food consumption have followed an overall GDP augmentation, exactly as observed for the per capita N consumption during the last few decades [*Billen et al.,* 2013; *Lassaletta et al.,* 2014a]. For a GDP range of $US 100–100,000, P consumption varies from 0.8 to 1.5 kg P capita^{-1} yr^{-1} to be compared with 2–7 kg N capita^{-1} yr^{-1} for proteins. In Vietnam, the Doi Moi policy in 1986 modernized the country and boosted the economy after a difficult recovery from the ravages of war. This can be clearly observed in the sharp increases in crop production, use of fertilizers, P consumption in the diet, and GDP from the 1990s (Figures 2 and 3).

Besides P fertilizers, net import of food and feed also accounts for new inputs of P. ASEAN-8 and EU-27 as well as the Red River Basin are net importers of P in agricultural products. In contrast, the Seine River Basin, with its fertile soils and its current specialization in cereal production, is exporting a large amount of P, over 900 kg P km^{-2} yr^{-1}. However, due to its very low livestock production, the Seine Basin has to import about 100 kg P km^{-2} yr^{-1} of animal products to meet the requirements of the local population. Considering a vegetal to animal conversion of 12–15%, the import of animal products into the Seine Basin virtually represents approximately two thirds of the net export of P as cereals.

4.1.3. Domestic Waste

While the fate of animal excretion is to be recycled onto agricultural land, that of human excretion differs between ASEAN-8 and EU-27. With only 35% connection to sewage facilities in ASEAN-8, a large share of domestic wastewater is recycled in agriculture, most of the remainder reaching surface waters. In EU-27, 34% of the P load from WWTPs is also recycled in agriculture, but 47% is retained in the WWTP and landfilled.

In the Red River, due to a lack of significant wastewater treatment before the very recent years, and taking into account differences in wastewater fate in rural and urban areas, we estimated the P reused for agriculture at 65%. For the Seine basin, from 80 kg P km^{-2} yr^{-1} in detergents plus the 300 kg P km^{-2} yr^{-1} from the human excretion, despite a 40% reuse of secondary sludge for agriculture (consistent with *Senthilkumar et al.* [2014]), only 60 kg P km^{-2} yr^{-1}, i.e., 16% of total P release, is recycled to agricultural land (compared to 65% in the Red River). Among the 230 kg P km^{-2} yr^{-1} of primary sludge immobilized in WWTP, about half is incinerated, the rest being disposed of in landfills [*IAURIF*, 2003]. Regarding, the amount of P in detergents, which still totals 20% of the total domestic waste, it should further decrease in the near future due to the regulation of dishwashing detergents [*Official Journal of the European Union Regulation*, 2012]. The proportion discharged into the river, amounting to about 20% of the raw effluent in the 2000s, has been further reduced to approximately 10% in the 2010s, so that an equal share between diffuse and point P sources is now observed [*Garnier et al.,* 2014]. Most industries efficiently treat their own effluents.

4.1.4. P Accumulation in Soils and Losses

The pool of P in agricultural soils is the key point of the whole P budget. Due to P's propensity to sorb on particles and form insoluble minerals, P has accumulated especially in Europe's overfertilized soils. The total P stored in the upper layer of the soil is on the order of 200 and 1000 10^3 kg P km^{-2}, respectively, for ASEAN and Europe [*Yang et al.,* 2013], so that any imbalance between input and output is difficult to detect over the short term. In France however, measurements show that the P content of soils increased by 15% between 1980 and the 2000s [*Lemercier et al.,* 2008]. For this reason, with the recent reduction in P inputs (technical agricultural institutes recommend not using P fertilizers systematically), crop production has not been reduced.

The EU-27 P soil budget shows an annual accumulation of 260 kg P km^{-2} for EU-27, but much less (60 kg P km^{-2}) for the Seine Basin, whereas in ASEAN-8 our budget indicates a net P soil removal over 114 kg P km^{-2} yr^{-1}, as a result of both lower inputs of P as fertilizers and manure, and higher erosion rates. In the Red River Basin, characterized by much higher P fertilizer application rates, rapid P accumulation in soil is occurring (332 kg P km^{-2} yr^{-1}). P from rocks weathering is low in these soils surface balances.

Regarding erosion, despite the problems assessing its rate [*SCU,* 2013], the erosion rates provided by *Valentin et al*. [2014] for several countries in ASEAN-8 and by *Cerdan et al.* [2010] for EU-27 gave for the Red River Basin similar figures to those calculated by *Le et al.* [2005] and *Luu et al.* [2012]. For the Seine watershed we found a rate of P loss by erosion (160 kg P km^{-2} yr^{-1}) higher than the previous estimates by *Némery and Garnier* [2007a]. This discrepancy results from the difficulty in determining whether eroded P-rich soil lost from crop fields reaches the drainage network, because it can be trapped in the downslope of cultivated soils or in artificially managed river wetlands [*Fiener et al.,* 2005; *Némery and Garnier*, 2007a; *Schoumans et al.,* 2013].

At the world scale, according to *MacDonald et al.* [2011], besides Europe, the coastal United States and southern Brazil show the highest P surpluses, together with Eastern Asia (notably China) [see *Han et al.,* 2013], whereas other places in the world have less access to fertilizer use, hence much lower surpluses or even strong deficits.

4.2. Environmental Threats

In European countries, eutrophication was combatted in lakes as early as the 1970s [*Vollenweider*, 1968], but the problem has spread extensively to rivers and coastal waters [*Cugier et al.,* 2005; *Garnier et al.,* 2010; *Passy et al.,* 2013; *Pistocchi et al.,* 2012; *Rabalais et al.,* 2009; *Romero et al.,* 2013] and can be a serious threat to ecosystem services (drinking water production, fisheries, and tourism) [*Lancelot et al.,* 2011]. In the 1970s, the P flux in rivers was dominated by the fraction originating from WWTPs, when the collection of domestic sewage increased but without adequate technology to remove P. Due to massive effluent discharges, the peak of disruption of the ecological functioning of lakes and reservoirs was reached in the 1980s. Awareness of water quality degradation in streams and rivers came later. In 1984, the Oslo-Paris convention (OSPAR) clearly considered P (and N) as water pollutants, and policies to reduce P started to be implemented in developed countries. One of the most important measures in Europe was the adoption of the Urban Waste Water Treatment Directive in 1991 and the banishment of phosphates in laundry detergents in some countries [*Billen et al.,* 1999; *Van Drecht et al.,* 2009]. France was one of the last countries to apply this measure in early 1990. Today, P is efficiently removed in WWTPs and has become the limiting nutrient in coastal waters (with regard to N and Si) in many areas of Western Europe. For the Seine Basin, P deliveries to the Seine Bight have been reduced by nearly a factor of 3 since the OSPAR convention in 1984, without a significant reduction in N [*Passy et al.,* 2013; *Romero et al.,* 2013]. With a more rapid decrease in P input to hydrosystems with regard to N, a nutrient imbalance (N $\gg$ P) has been observed in EU-27 river deliveries to coastal waters [*Billen et al.,* 2011; *Bouraoui and Grizzetti*, 2011; *Grizzetti et al.,* 2012; *Romero et al.,* 2013] and specifically on the French and Belgian North Sea coast [*Billen et al.,* 2007; *Passy et al.,* 2013; *Thieu et al.,* 2009]. Harmful or toxic algal blooms regularly create severe problems, including fish contamination leading to fishery prohibition. Several studies have shown that a drastic change in agricultural systems is required to return to a nutrient balance by lowering N compared to P [*Thieu et al.,* 2011; *Passy et al.,* 2013]. Despite the general abatement of P in the surface waters of European countries, depending on the soil characteristics, the topography, and the human activities in the watershed, part of the P accumulated during decades in European soils can still be lost and transferred to the river system [*Cordell et al.,* 2009; *Ekholm and Lehtoranta*, 2012; *Quinton et al.,* 2010; *Sattari et al.,* 2012].

For Asian developing countries, the mechanisms of P transfer to the hydrosystem differ due to (i) the higher erosion propensity in subtropical monsoon rivers, among which major ones are coming from Himalaya mountains [*Valentin et al.,* 2008] and (ii) the low level of sanitation [*Montangero et al.,* 2007; *Van Drecht et al*., 2009]. P river loads are indeed represented by a high proportion of diffuse P (agricultural soils especially). With 35% connection to sewage in the 2000s, human excreta are still more delivered on cultivated land than directly in surface waters. However, with the rapid growth of cities, domestic effluents in developing countries are becoming a real challenge for water quality [*Morée et al.,* 2013]. In the future, most of the domestic sewage of Asian cities will be collected in WWTPs. Without appropriate treatment, a large proportion of the collected wastewater will be discharged into the rivers, and the accompanying excess P could lead to eutrophication and severe harmful algal blooms (toxic, mucilaginous, etc.) in the rivers and their connected stagnant systems, as well as in coastal zones [*Heisler et al.,* 2008]. The situation is evolving quickly in Asian countries, especially in China where harmful algal blooms are already currently observed [*Heisler et al.,* 2008; *Liu et al.,* 2013; *Xinyan et al.,* 2011; *Strokal et al.,* 2014]. Considering *Prorocentrum minimum* as a good indicator of eutrophication in the world, *Heisler et al.* [2008] showed that ASEAN-8 countries are not yet encountering high eutrophication, as shown for the Chinese and European coastal areas. For example, N deliveries from the Red River to the coast have increased, while P deliveries have remained stable, both staying close to equilibrium with Si, so that the risk of eutrophication at the coast is still limited [*Garnier et al.,* 2010; *Le et al.,* 2014]. However, eutrophication including the development of harmful algal blooms could rapidly increase due to an increasing urban population and wastewater collection systems discharging directly into the rivers.

4.3. Improving the Management of Phosphorus

To improve P management at the regional and basin scale, more attention would be needed on allowable emissions into waterways and thus saving this limited resource for agriculture [*Linderholm et al.,* 2012; *Smil*, 2011]. In this respect, P differs from fossil C and industrial N fertilizers, which can have substitutes, such as renewable energy sources or N fixation by legumes, respectively. P is derived from a limited minable resource stock and only parsimonious use and recycling can postpone its exhaustion. Good management of P therefore concerns the entire water-agro-food system.

4.3.1. At Macroregion and National Scales

Diet has already been shown to be a strong lever for reducing N contamination in waters [*Billen et al.,* 2012; *Lassaletta et al.,* 2014b]. Similarly for P, decreasing meat consumption by half in developed countries, while offering the opportunity to increase meat consumption in developing countries to that same level, would limit increased crop demand (especially crops used for feeding animals) and fertilizer use as well. This would involve a geographical redistribution of resources more than a global reduction [*Billen et al.,* 2015]. Based on material flow analysis, several local studies have indeed shown a 20–45% decrease in the P demand of fertilizers for a vegetarian diet [*Schmid-Neset et al.,* 2005] (*Tangsubkul et al.* [2005], cited in *Cordell et al.* [2009]). To address the effects of dietary choices on P pollution, new efforts would be needed in childhood education, media communication, and local organizations regarding meat and milk consumption, not only for resource sustainability [*Metson et al.,* 2012] but also for health reasons [*Westhoek et al.,* 2014].

Together with diet, agricultural production system is a real challenge. In EU-27 and especially in France, opportunities exist to reconnect farming and cattle breeding to benefit from animal excretion for fertilization and for a relocalization of food processing and consumption. In ASEAN-8, preventing a possible P shortage in the future may require retaining current connections between crop and animal production systems.

Regarding P domestic wastewater in most countries of EU-27, the extent of their collection without appropriate treatment in wastewater treatment plants in the 1960–1990 period has caused eutrophication of most water masses, a lesson that could be taken now in ASEAN-8 where sanitation still low, is in rapid progress. Therefore, P banishment from detergent remains necessary, as a curative measure in EU-27 and a preventive one in ASEAN-8. The acceptability of this measure is much higher for the market if the entire region adopts the same regulation, stimulating the search for competitive alternative solutions. More recently, P recycling policies have been planned and coordinated at the national or regional level for EU-27 with the very recent Commission Communication on the circular economy (COM COM (2014) 398 final/2 25.9.2014), which mentions the need for improving P efficiency. ASEAN-8 countries, where P is still recycled at a much higher rate, have the opportunity to anticipate the systematic P recycling from WWTPs sludge and thus avoid some environmental damages already experienced in EU-27.

4.3.2. At the River Basin Scale

Interestingly, most of the policies planned at the macroregion and national scales are implemented by river basins. For historical reasons related to water availability and the role of waterways in commodity transportation, the largest cities are generally located along the lowest sectors of the river networks and the upstream basin has often provided a large share of its resources, including food, energy, and building materials, to downstream densely urbanized areas [*Barles*, 2008; *Billen et al.,* 2012].

Regarding agricultural food production, wastes produced by cities (green waste, human and animal excreta, etc.) were traditionally returned to rural soils. However, in the last 50 years, these connections have been progressively lost in the Seine Basin. Due to the demographic increase of the Paris conurbation, representing two thirds of the Seine Basin population, the intensive industrial agriculture relocated the rural population to urban areas. Excessive fertilization in the Seine Basin has resulted in high soil P content. The general recommendation is avoiding soil P losses. Several agricultural practices have proved to be efficient in keeping P in soils. Plowing straw, incorporating manure into soils, using cover crops and/or crop associations, preventing erosion, etc. [*Schoumans et al.,* 2013; *Tiecher et al.,* 2012] are all measures that can be generalized. Such practices are also appropriate for N fertilization (e.g., avoiding NH_3 volatilization from urea application [*Sanz-Cobena et al.,* 2014], reducing NO_3 leaching [*Askegaard et al.,* 2011], and enriching soils in organic matter and improving water retention [*Tiecher et al.,* 2012]). In addition, because agriculture intensification in the Seine Basin also led to specialization in cash crop production (cereals), while cattle breeding strongly decreased, moving to other regions specializing in livestock raising, a reconnection of livestock with cropping systems can make manure available to soils while limiting transport [*Asai et al.,* 2014; *Bonaudo et al.,* 2014; *Herrero et al.,* 2010; *Lassaletta et al.,* 2014a]. Regarding P availability to plants, new variety selections and the association of crops with mycorrhizal fungi are new ways to make phosphate available to crops in future sustainable agriculture without increasing fertilizer inputs [*Betencourt et al.,* 2012].

In the Red River Basin, the situation of the 2000s was different due to the more intensive agricultural areas situated in the delta (and not in the upstream basin as in the Seine). Furthermore, the delta covers a relatively small surface (14,300 km^2), with rice as the main crop, managed on small family plots together with some cattle. Similarly, gardening was common in the vicinity of Hanoi, providing fresh vegetables to the urban population. However, in the last few years, the delta's population has moved to the largest cities, especially Hanoi, which extended its administrative limits to Hay Tay province in 2008. Moreover, due to hard working conditions, a shift from rice cultivation to aquaculture is a current trend of the agricultural system in the Red River delta. Fish ponds have been shown to be a potential source of eutrophication via nutrient mineralization [*Bouwman et al.,* 2013]. Because of ongoing radical changes in the Red River Basin's agricultural system, a diversification of complementary agricultural activities (e.g., aquaponics; reuse of domestic effluent; and combining crop, livestock, and fish farming) would be beneficial for lowering the cost of production by reducing the amount of fertilizer and preventing environmental damage. An important issue is also the proper management of manure in pastures, as *Sattari* [2014] has recently shown in the risk of fertility transfer from grassland to cropland. In the upstream hillslopes of the Red River Basin, the major threat is erosion, which could be accentuated by deforestation and expansion of agriculture land, although the link between erosion, climate, and land use is not linear [*Dotterweich*, 2013].

Due to pedoclimatic and cultural specificities, agricultural systems and practices, although today strongly dependent on a globalized market, call for strong structural changes adapted to more regional constraints [*Hinrichs*, 2013]. Therefore, in order to reconcile agriculture production and a high-quality environment, the watershed is a good scale of management, especially in terms of water quality (eutrophication and domestic use) and quantity (agriculture irrigation).

Regarding wastewater, its management also plays a key role in the environmental impacts of P and on the efficient use of the P resource, because cities represent a potentially considerable source of P that can be recovered and reused [*Cordell et al.,* 2011; *Günther*, 1997; *Muster et al.,* 2013]. The ban of P in detergents has successfully reduced surface water contamination significantly (e.g., in Europe and specifically in the Seine Basin [*Garnier et al.,* 2005; *Passy et al.,* 2013]) and a similar benefit could be expected if such policies were adopted in Asia. Human excreta are a potential P source, together with other solid wastes shown to contain large amounts of P that cannot be neglected [*Kalmykova et al.,* 2012; *Ott and Rechberger*, 2012].

The largest wastewater treatment plants of the Seine Basin, located in the Parisian conurbation, have reached their limit of development (both in technical and in land use terms) with the application of the Urban Waste Water Directive [*Urban Waste Water Treatment Directive*, 1991]. In the context of a future scenario of expansion of Paris toward the sea (cf. Le Grand Paris [*Attali*, 2010]), innovative management of domestic wastewater may be needed. Considering the continuing population migration to large cities (from Paris to Le Havre), the recovery of human excreta would tighten P cycle, although high water content in organic wastes might makes this recovery problematic [*Kirchman et al.,* 2005]. At present there is a variety of end-of-pipe technologies allowing the recovery and reconcentration of P in bioavailable forms [see *Muster et al.,* 2013] or recovery after incineration [see *Nanzer et al.,* 2014]. However, source separation and recycling of excreta fractions as fertilizers to agriculture have been shown to have a good potential for protecting the environment [*Antonini et al.,* 2011; *Libralato et al.,* 2012; *Spångberg et al.,* 2014]. In addition, for a megacity such as Paris, P related to both food waste [*Grizzetti et al.,* 2013] and other solid wastes can be advantageously reused for energy provision and secondary resource recovery [*Astrup et al.,* 2014]. Moreover, sewage from animal production and food-processing industries are potential sources of P that could be fully inventoried for better recovery. These are typical actions that can be managed at the territorial scale of cities and watersheds, with an integrative view of the water-agro-food systems.

In the Red River delta, Hanoi, now reaching 7 million inhabitants and facing wastewater problems possibly affecting the economic development of the city, constructed WWTPs (the Yen So Park WWTP being the largest, designed for 200,000 $m^3 d^{-1}$, i.e., about 2 million inhabitants) are not fully operating at present. Other scheduled WWTP construction will further treat 200,000 $m^3 d^{-1}$. In addition to reuse of sludge in agriculture, a major recommendation would be to ensure the operability of the plants while collecting wastewater. Regarding sanitation in Paris, we have shown that the highest level of surface water pollution occurred when a large fraction of collected domestic wastewater was discharged into the river with no or poor treatment [*Billen et al.,* 2007]. It should be mentioned, however, that the Red River itself has escaped eutrophication because of its high turbidity, limiting algal growth, as well as strong organic contamination, due to dilution stemming from high discharge. Nevertheless, urban rivers around Hanoi are heavily organically polluted and/or eutrophicated [*Anh et al.,* 2006], being used as open sewers, as was the case in Paris or Brussels before efficient sanitation [*Billen et al.,* 1999; *Garnier et al.,* 2013]. However, while sanitation is progressing, it is time to plan for a source separation and recycling of excreta fractions as fertilizers to agriculture. Such practices are still common in the rural and periurban areas of developing Asian countries, and simple technical skills could easily solve the currently associated hygiene and health problems [*Heinonen-Tanski and Van Wijk-Sijbesma*, 2005].

According to *Smil* [2000], a P resource that should be considered in this basin is the reuse of P accumulated in eutrophicated urban water bodies (bottom sediments of rivers and ponds), and the P accumulated in reservoirs rapidly filled by erosion and sedimentation of P-rich suspended solids [*Dang et al.,* 2010; *Quinton et al.,* 2010; *Vörösmarty et al.,* 2003]. Finally, in developing countries such as Vietnam and its populated deltas, the increase of P consumption per capita with increasing GDP is likely to result in an increase in solid wastes—exactly as occurred in Europe in the past (+54% for EU-15 during the 1980–2005 period)—which can be anticipated [*Kalmykova et al.,* 2012].

5. Conclusion

Taking into account the specificities and the socioecological trajectories of the two regions analyzed in this study, P fluxes show a different status. The increase in P fluxes of ASEAN-8 is associated with both agricultural intensification and agricultural expansion, while in Europe the trajectories of P fluxes are associated with improvements in agricultural performance. After a drastic P reduction in surface waters, EU-27 countries are now aiming at reducing the P soil surplus and at recovering P from sewage and wastes, while ASEAN-8 countries are working toward improving sanitation by collecting and treating wastewaters. This in turn could rapidly damage the quality of large rivers and coastal zones if the appropriate targets of P abatement are not reached. Considering the scarcity of P resources, the continuously increasing use of P fertilizers in ASEAN-8 agriculture could accentuate the growing gap in access to P fertilizers between developed and developing countries. For better sustainability of P fertilization, nutrient management at different organizational levels must be considered, including policies to improve reuse of P waste at the national and regional levels and good practices at the farm level, which take into account soil properties (natural P fertility and erosion risks) and climate conditions [*MacDonald et al.,* 2011; *Senthilkumar et al.,* 2012].

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