



1 **Global and regional phosphorus budgets in agricultural systems and their**
2 **implications for phosphorus-use efficiency**

3 Fei LUN ¹, Junguo LIU ^{2*}, Philippe CIAIS ³, Thomas NESME ⁴, Jinfeng CHANG ³,
4 Rong WANG ³, Daniel GOLL ³, Jordi SARDANS ⁵, Josep PEÑUELAS ⁶, Michael
5 OBERSTEINER ⁷

6 1. College of Resources and Environment Sciences, China Agricultural University,
7 Beijing 100193

8 2. School of Environmental Science and Engineering, South University of Science and
9 Technology of China, Shenzhen, 518055, China

10 3. IPSL – LSCE, CEA CNRS UVSQ, Centre d'Etudes Orme des Merisiers, 91191 Gif-
11 sur-Yvette, France

12 4. Bordeaux Sciences Agro, Université de Bordeaux, UMR 1391 ISPA, CS 40201,
13 33175 Gradignan Cedex, France

14 5. CREAF, Cerdanyola del Vallès 08193, Catalonia, Spain

15 6. CSIC, Global Ecology Unit CREAF-CSIC-UAB, Cerdanyola del Vallès 08193,
16 Catalonia, Spain

17 7. International Institute for Applied Systems Analysis, 2361 Laxenburg, Austria

18 * Author for correspondence: Junguo Liu. Phone/fax: +86-0755-88018012. Email:
19 liu_junguo@163.com

20 **Keywords:** phosphorus budget, phosphorus-use efficiency, global scale, regional scale,
21 country scale, agriculture

22



23 Abstract

24 The application of phosphorus (P) fertilizer to agricultural soils increased by 3.2%
 25 annually from 2002 to 2010. We quantified in detail the P inputs and outputs of
 26 cropland and pasture, and the P fluxes through human and livestock consumers of
 27 agricultural products, at global, regional, and national scales from 2002 to 2010.
 28 Globally, half of the total P input ($21.3 \text{ Tg P yr}^{-1}$) into agricultural systems accumulated
 29 in agricultural soils during this period, with the rest lost to bodies of water through
 30 complex flows. Global P accumulation in agricultural soil increased from 2002 to 2010,
 31 despite decreases in 2008 and 2009, and the P accumulation occurred primarily in
 32 cropland. Despite the global increase of soil P, 32% of the world's cropland and 43%
 33 of the pasture had soil P deficits. Increasing soil P deficits were found for African
 34 cropland, versus increasing P accumulation in Eastern Asia. European and North
 35 American pasture had a soil P deficit because continuous removal of biomass P by
 36 grazing exceeded P inputs. International trade played a significant role in P
 37 redistribution among countries through the flows of P in fertilizer and food among
 38 countries. Based on country-scale budgets and trends we propose policy options to
 39 potentially mitigate regional P imbalances in agricultural soils, particularly by
 40 optimizing the use of phosphate fertilizer and recycling of waste P. The trend of
 41 increasing consumption of livestock products will require more P inputs to the
 42 agricultural system, implying a low P-use efficiency aggravating the P stocks scarcity
 43 in the future. The global and regional phosphorus budgets and their PUEs in agricultural
 44 systems is publicly available at <https://doi.pangaea.de/10.1594/PANGAEA.875296>.



45 1. Introduction

46 Population increases and dietary changes require higher food production, which
47 increases global demand for fertilizers (Grote *et al.*, 2005; Foley *et al.*, 2011).
48 Phosphorus (P) is an essential element for all organisms, and a lack of P limits growth.
49 Fertilizer P enhances agricultural production, but P is also fixed in soils and can
50 accumulate. In countries with high fertilizer use, much P is lost to leaching and runoff,
51 leading to eutrophication of both inland and coastal waters (Carpenter *et al.*, 1998;
52 MacDonald *et al.*, 2011).

53 To supply the growing need for P in fertilizer, mining of phosphate rock has
54 quadrupled in the past half century, increasing from 46 Mt in 1961 to 198 Mt in 2011
55 (Scholz *et al.*, 2013). Despite some short-term fluctuations in the price of phosphate
56 rock, the global production of fertilizer P has been steadily increasing, at a rate of 3%
57 to 4% annually during the half century before 2011, and is projected to increase by 50
58 to 100% by 2050 (Cordell *et al.*, 2009, 2012). Extractable phosphate rock is a non-
59 renewable resource, and significant depletion of the resource is projected by the end of
60 this century if the current intensive use continues, possibly leading to resource shortages
61 (Cordell *et al.*, 2009; van Vuuren *et al.*, 2010; Peñuelas *et al.*, 2013).

62 The mining of P and its application as fertilizer in cultivated land is a major
63 anthropogenic perturbation of the natural biogeochemical P cycle (Carpenter and
64 Bennett, 2011; Elser and Bennett, 2011; Steffen *et al.*, 2015). The negative impacts of
65 this perturbation on the natural environment depend on how much P is lost from regions
66 with intensive fertilizer use (Smil, 2000; Bennett *et al.*, 2001).

67 P application differs significantly between countries and crop types (Grote *et al.*,
68 2005), and previous researchers have attempted to estimate the P flows in agricultural
69 systems in Europe (Ott & Rechberger, 2012), the United States (Suh & Yee, 2011),



China (Ma *et al.*, 2011), France (Senthilkumar *et al.*, 2012), Australia (Cordell *et al.*, 2013), and the world (Smil, 2000; Liu *et al.*, 2008; MacDonald *et al.*, 2011; Schipanski & Bennett, 2012). International trade and regional agricultural policies affect P budgets by increasing or decreasing the gap between P inputs and P outputs in agricultural land (Grote *et al.*, 2005). Previous research mainly focused on cropland while P fluxes in pasture and livestock production systems received less attention (McDowell and Condon, 2004) hampering the differences in methodologies, system boundaries, and data sources have made it difficult to assess the differences in the phosphorus use efficiencies among agricultural sectors and to extrapolate regional findings to the global scale.

To mitigate these problems, we (1) compiled a detailed and harmonized dataset of P fluxes in agriculture for countries around the world, including detailed analysis of input and output fluxes for cropland, managed grassland (hereafter, pasture), livestock, and human consumers of agricultural products; (2) characterized P budgets and P-use efficiencies in those different sub-systems; and (3) examined how international trade of phosphate fertilizer and agricultural commodities influences regional P fluxes. We performed this analysis at the scale of countries, regions, and the world; wherever possible, we distinguished different crop types. The study period was from 2002 to 2010, allowing us to study temporal trends.

2. Materials and methods

In this study, we obtained data for 224 countries (Table SI-1 in the supporting information). We defined the agriculture system as cropland and pasture ecosystems, plus human and livestock consumers of agricultural production and of other products containing P (Fig. 1). External P inputs to the agriculture system came from mined phosphate rock and atmospheric deposition. Several processes cause P losses from the



95 system into the external environment (here, defined as non-agricultural land and bodies
96 of water). Figure 1 presents the fluxes of P into and out of the agriculture system at a
97 global scale, including internal fluxes between ecosystems and consumers. We
98 quantified these fluxes in the present study based on a mass-balance approach (Cordell
99 *et al.*, 2012). We defined the phosphorus-use efficiency (PUE) of the agricultural
100 system and of its subsystems as the ratio of the total P harvested in economic outputs
101 (e.g., crops, meat, milk and eggs) to the total P input. International trade in fertilizer
102 and food is discussed separately in section 2.3. The data sources and an overview of the
103 mass-balance equations are presented in the rest of this section; details and equations
104 are presented in the Supporting Information (SI).

105 **2.1 P flows into and out of the agricultural system**

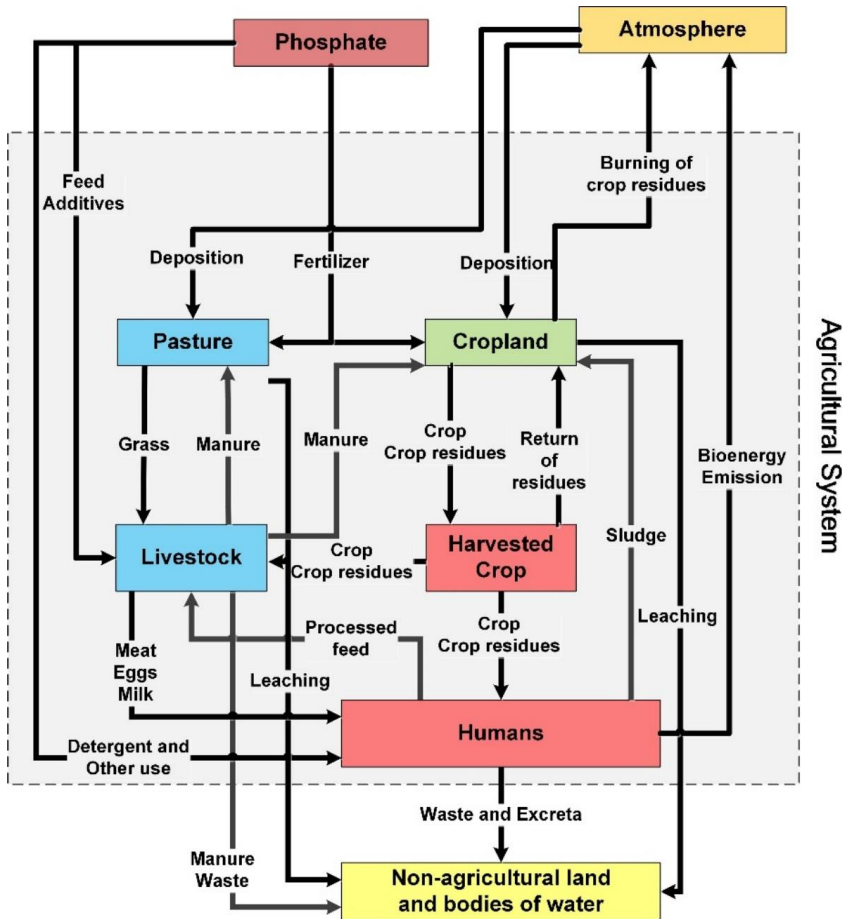
106 Inputs into the agricultural system, which is within the gray box in Fig. 1, are from
107 mined phosphate rocks and atmospheric deposition. We did not include P from in situ
108 weathering of soil particles because the rate of this process is insignificant compared
109 with the magnitude of other inputs (Liu *et al.*, 2008). Outputs included P emission into
110 the atmosphere from fires and P loss to uncultivated land or bodies of water.

111 **2.1.1 P inputs**

112 Data on agricultural inputs of phosphate P in fertilizers were collected from the
113 International Fertilizer Industry Association (<http://www.fertilizer.org>) and divided
114 between cropland and pasture uses based on information from FAO (2002) and the
115 FAOSTAT database (<http://www.fao.org/faostat/en/#data>). A small fraction (8%) of P
116 from mined phosphate rock is used to produce animal feed additives. Apart from
117 fertilizer and animal feed additives, the rest of the mined P is used to produce detergents
118 and other products directly consumed by humans (Ringeval *et al.*, 2014). Atmospheric
119 P deposition in cropland and pasture areas was calculated separately in each country



120 using gridded global P-deposition maps obtained using the LMDz-INCA aerosol
 121 chemistry transport model of Wang *et al.* (2014, 2015) and agricultural land-use maps.
 122 Details are provided in the Supporting Information (Table SI-2).



123
 124 Figure 1: Scheme of the P pools and fluxes used to diagnose global P budgets for the
 125 agricultural sector. The agricultural sector (or system) in the grey box includes cropland
 126 and pasture soils, livestock, human consumers of livestock and crop products and users
 127 of phosphate derived products. National and regional P budgets are calculated using the
 128 same scheme, but including in addition exports and imports of P embedded in traded
 129 crop and livestock products, and fertilizers.



2.1.2 P outputs

P emission from agricultural fires was obtained from the gridded dataset of Wang *et al.* (2015), and cover the burning of crop residues in the field, by households, and for the production of bioenergy from crop biomass. Leaching from cropland and pasture soils was assumed to be a constant fraction (12.5%) of P inputs for each agricultural land use type (Bouwman *et al.* 2013). P outputs from non-recycled livestock and human manure were calculated based on the mass balance. Note that erosion-induced losses of P are important in many agricultural regions (Quinton *et al.*, 2010), but were not considered in this study because we lack data on the re-deposition of P in eroded soil material from agricultural soils. In future research, it will be important to quantify this source of P, particularly in agricultural areas that receive large annual inputs of sediment (e.g., in river floodplains and sites on steep terrain that experience significant erosion farther up the slope followed by deposition).

2.2 P flows within the agricultural system

2.2.1 P in harvested crop biomass and crop residues

The flux of P in harvested crop biomass was estimated from yield data (FAOSTAT) using crop-specific P concentrations, after grouping 178 different crops into 13 crop types (COMIFER, 2007; USDA-NRCS, 2009; Waller, 2010; Table SI-2). P in harvested crop biomass was partitioned into crops (for human and livestock consumption) and crop residues (Fig. 1). We estimated the P fluxes of crop residues from FAOSTAT data and from Liu *et al.* (2008) to account for residue that is recycled in the field (50%), transformed into livestock feed (25%), and burned or used by other human activities (25%).

2.2.2 P in grazed biomass

The P removed from pasture by livestock grazing was estimated by combining



155 forage grass consumption data with the P concentrations in grass biomass (Antikainen
156 *et al.*, 2005; COMIFER, 2007; USDA-NRCS, 2009; Waller, 2010). Gridded data on
157 grass biomass consumption by livestock were obtained by combining the global
158 livestock production systems dataset of Herrero *et al.* (2013) with pasture net primary
159 productivity simulated by the ORCHIDEE-GM global pasture model (Chang *et al.*,
160 2013, 2015). We chose the ORCHIDEE-GM model for this analysis because it is able
161 to separate the intake of grazed vs. cut forage grass.

162 **2.2.3 P in animal feed products**

163 Animal feed products used as complementary diet (“feed additives”) represent
164 direct inputs to the livestock sub-system (Fig. 1). This flux was deduced from the mass
165 balance of the known input and output fluxes for the livestock P pool, but did not
166 account for long-term changes in P storage in that pool. See the Supporting Information
167 for more details.

168 **2.2.4 P embedded in livestock products**

169 This flux of P leaving the livestock subsystem and entering the human subsystem
170 (Fig. 1) through the harvesting of products was calculated by multiplying the
171 FAOSTAT production data for meat, eggs, and milk by the product-specific P
172 concentrations reported by Grote *et al.* (2005).

173 **2.2.5 P in livestock manure**

174 We calculated the manure P production based on FAOSTAT data about N in
175 livestock manure and P:N values for each types of livestock manure (MWPS-18, 1985;
176 OECD Secretariat, 1991; Levington Agriculture, 1997; Sheldrick *et al.*, 2003; ASAE,
177 2005) (see in the Table SI-3). Once produced, manure P is either applied to cropland,
178 left in the pasture, or lost to the environment as waste (Fig. 1), following the same
179 partitioning as that for N in the manure from FAOSTAT.



2.2.6 P in human sewage sludge

We assumed that the P output from humans equaled the inputs from non-fertilizer P ore products and the consumed crop and livestock products (Fig. 1), and used this to calculate the total P production in human excreta. P in human sewage sludge was estimated using population data and values of per capita production of P in excreta (Smil, 2000; Cordell *et al.*, 2009). Following the method of Liu *et al.* (2008), we assumed that 30% of the excreta P from urban populations and 70% of P from rural populations were returned to cropland, either directly or after treatment of sewage sludge, with the remaining P assumed to be lost to the environment (e.g., in landfills or bodies of water).

2.3 P flows from international trade

We compiled the flows of P in international trade both from the P embodied in crops and livestock products and in P embodied in fertilizers exchanged between countries. For agricultural commodities, we used FAOSTAT data that provided a matrix of commodities exchanged between countries, and converted this data into P fluxes using commodity-specific P content data. For P fertilizers, we used the International Fertilizer Industry Association trade statistics. By convention, a positive trade balance for a country means that it is a net P importer. In addition, P fluxes associated with the international trade of fertilizers, food, feed, and fiber commodities can be associated with local cropland PUE and pasture PUE. We defined the dependency on fertilizer imports (F_{fer}) as the ratio of the P in imported fertilizers ($P_{\text{fer-imp}}$) to the P in all fertilizers consumed by a country ($P_{\text{fer-con}}$). Similarly, we defined the dependency on food imports (F_{food}) as the ratio of P in food imports ($P_{\text{food-imp}}$) to the P in all food consumed by a country. Furthermore, we defined F_{total} as the ratio of the total P imported (food and fertilizers) to the total P consumed as fertilizers and food in



a country. The equations for these calculations are presented in sections 2 to 6 of the Supporting Information.

2.4 Annual P budgets of cropland and pasture soils

Annual changes in P stocks in cropland and pasture soils (ΔP) were estimated as the difference between inputs and outputs (i.e., the budget); $\Delta P > 0$ indicates net P accumulation in the soil, $\Delta P < 0$ indicates a net deficit, and $\Delta P = 0$ represents no net change. ΔP calculated in this manner does not reflect the legacy effects from previous management and fertilization practices (Ringeval *et al.*, 2014), but it is a useful metric to identify regions with a P surplus or deficit at any point in time and to compare countries.

Annual soil ΔP values were calculated as the differences between annual inputs and outputs. Details and the equations are presented in section 2 of the Supporting Information.

2.5 Cumulative P budgets of cropland and pasture soils

Following the method of Sattari *et al.* (2012), we separated the P inputs to soils (except inputs in seeds) into two pools: (1) a stable P pool, which represents P that is unavailable to plants on an annual basis, such as the P absorbed onto iron and aluminum oxides (20% of total P inputs, including fertilizers, manure, sludge and deposition); and (2) a labile P pool that is assumed to be available for plant uptake (80% of total P inputs). P can be exchanged between the two pools. If inputs of labile P are larger than P removal in crop biomass, we assumed that the surplus labile P gets transferred into the stable P pool at the end of the year. In the opposite case, in which inputs of labile P are lower than P removal, plants can take up P from the stable pool (Sattari *et al.*, 2012). This approach assumes that the P loss by runoff and leaching into bodies of water is from the labile P pool only, and that P stored in seeds does not belong to either the



230 stable pool or the labile pool. This approach is simplistic, as more research will be
231 required to allow a more realistic modeling of these two pools and of the flows they are
232 involved in.

233 **2.6 Phosphorus-use efficiency**

234 We defined PUE as the ratio of P in the harvested economic outputs to P in the
235 inputs for the entire agricultural system (the gray area in Fig. 1) or for a given subsystem.
236 PUE indicates how much of the input P is transferred into value-added products. If
237 $PUE > 1$, the input of P is insufficient to sustain the output (harvested P), suggesting a
238 net reduction of the system's P reservoir. For cropland PUE, we defined P in harvested
239 crops as the economic P output of the crops, and the sum of phosphate fertilizer,
240 livestock manure, human sewage sludge, and P from atmospheric deposition as the P
241 input. For pasture PUE, harvested P refers to the P consumed by grazing animals, and
242 the sum of phosphate fertilizer, livestock manure going to the pasture, and P from
243 atmospheric deposition as the total inputs. For the livestock subsystem, the harvested P
244 output represents the P in livestock products (meat, eggs, and milk), whereas the inputs
245 represent the input into livestock. We also defined the PUE of human food (ϵ_{food}) as the
246 ratio of the P content in human excreta to the total P input in human food; this represents
247 an inconsistency with our previous definitions, since human excreta have currently no
248 economic value. The equations for all the PUE terms are provided in section 5 of the
249 Supporting Information.

250 **2.7 Uncertainty estimates**

251 Uncertainties in each flux originate both from the material flux data and from data
252 on the P concentration in each material considered by our analysis, including crop
253 products, crop residues, livestock, meat, eggs, milk, livestock, and human excreta.
254 Many of the global statistical datasets used in our analysis are not replicated, and no



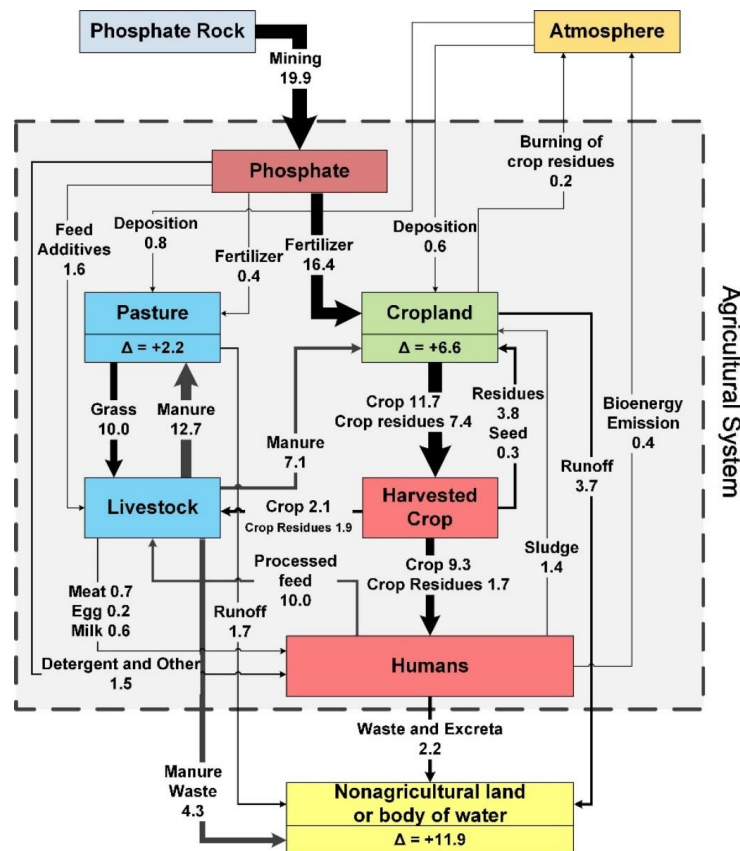
255 alternative dataset is available for establishing a range of uncertainty values for the
256 different P fluxes. National datasets have usually not been formally analyzed to
257 determine their uncertainty, and many of the sources of uncertainty are difficult to trace
258 (e.g., clerical errors, differences between countries in product definitions). Thus, we
259 have only addressed the effect of uncertainties in the P concentration by means of
260 Monte Carlo simulations (3000 iterations) using the range of P concentrations reported
261 in the literature (Table SI-5).

262 3. Results

263 3.1 Global agricultural P flows and their trends

264 3.1.1 Global P fluxes in and out of the agricultural system

265 Figure 2 summarizes the annual average of global P flows for the period from 2002
266 to 2010. P from phosphate fertilizers was the largest single input flux, representing 93%
267 of the 21.3 Tg P yr⁻¹ of the global input, and most of it (82.4%) goes to cropland and
268 pasture. Outputs from the agriculture system amounted to 12.5 Tg P yr⁻¹, which
269 combines outputs from leaching and runoff into bodies of water (5.4), non-recycled
270 manure waste (4.3) and sewage (2.2), bio-energy (0.4), and burned crop residues (0.2).
271 The global annual P balance of agricultural systems was therefore positive during the
272 entire study period, with 8.8 Tg P yr⁻¹ accumulating in soil, of which 6.6 Tg P yr⁻¹
273 accumulated in cropland and 2.2 Tg P yr⁻¹ in pasture. On average, 41% of the P input
274 accumulated in soils from 2002 to 2010.



275

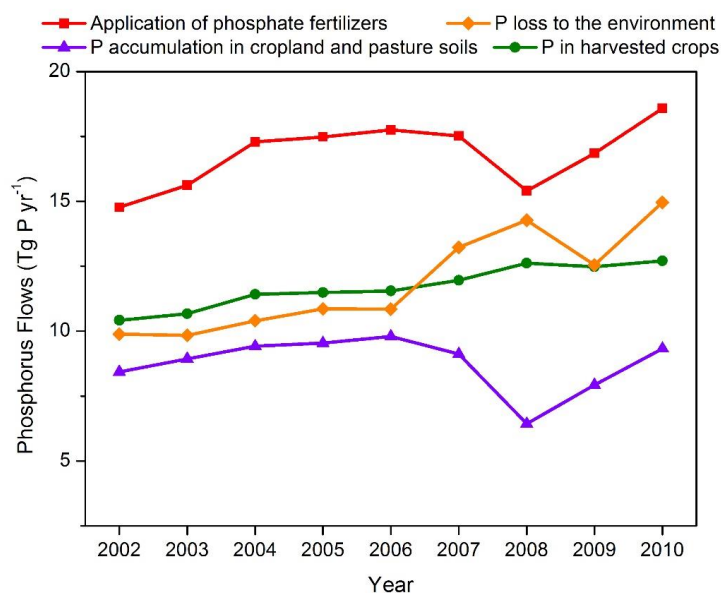
276 Figure 2: Annual P flows in the global agriculture system from 2002 to 2010. Values
 277 are Tg P yr⁻¹. The notation Δ denotes the average change of P in pasture and cropland
 278 soils, respectively. By convention, a positive value means accumulation. Note that
 279 livestock and humans changes of P are assumed to be zero.

280 3.1.2 Temporal trends

281 Figure 3 shows the trends for the four largest P fluxes in the agriculture system.
 282 Application of phosphate fertilizer increased at an average annual rate of 3.2% from
 283 2002 to 2010, despite a decrease in 2008 that reflected reduced fertilizer application at
 284 a time when the price of phosphate fertilizers increased (Cordell *et al.*, 2009, 2012).
 285 The trend for P in harvested crop biomass was also a steady increase, but at a lower
 286 annual rate (2.4%) and with no decrease in 2008, probably because of the availability



287 of P that accumulated in the soil from previous years (as described in section 2.5).
288 Overall, P in agricultural soils increased by 1.3% annually, whereas P losses to the
289 environment increased faster ($6.4\% \text{ yr}^{-1}$) than fertilizer inputs.



290
291 Figure 3: Time series in the four largest global annual P flows, within, in and out of the
292 agriculture system from 2002 to 2010.

293 3.1.3 Global P fluxes in cropland

294 Cropland received both the largest fraction (82%) of phosphate fertilizer and 29%
295 of the manure produced by livestock, as well as all of the recycled human sewage sludge
296 (Fig. 2). Atmospheric deposition contributed an additional 0.6 Tg P yr^{-1} of inputs to
297 cropland. Harvesting of cropland removed $11.7 \text{ Tg P yr}^{-1}$, which can be divided into
298 crop products used for human nutrition (9.3 Tg P yr^{-1} , including 5.3 for food, 2.7 for
299 processing, 0.4 for waste and 0.9 for other use) and for livestock feed (2.1 Tg P yr^{-1}),
300 with a small pool in seeds returned to the cropland (0.3 Tg P yr^{-1}). On average, 50% of
301 the P contained in crop residues was recycled to cropland during the study period, with



0.2 Tg P yr⁻¹ lost to the atmosphere from burning of crop residues. The remaining 3.6 Tg P yr⁻¹ contained in harvested crop residues is removed from cropland and redistributed to livestock and humans. Globally, 3.7 Tg P yr⁻¹ was lost from cropland soils through leaching and runoff. The sum of all these fluxes results in an annual soil P accumulation of 6.6 Tg P yr⁻¹ (Fig. 2).

The global cropland PUE averaged 0.46, with a maximum of 0.51 in 2008 and a minimum of 0.44 in 2006. The annual cropland P accumulation ratio (cropland soil P accumulation / total P input to cropland) was 23%, which is lower than the accumulation ratio of 48% found for the overall agriculture system. In countries where labile P inputs were lower than P removal in crops, the soil's labile P pool was depleted by 1.9 Tg P yr⁻¹ by harvesting of crop biomass. In countries where labile P inputs are higher than P removal by crops, the accumulation of soil labile P was 6.0 Tg P yr⁻¹. Thus, there is an asymmetry between these two groups of countries, with accumulation being larger than depletion at a global scale. In addition, the global stable P pool in cropland increased by an average of 5.6 Tg P yr⁻¹ from 2002 to 2010.

3.1.4 Global P fluxes in pasture

Figure 2 shows that most P inputs to pasture were from livestock manure (12.7 Tg P yr⁻¹), with small additional contributions from atmospheric deposition (0.8 Tg P yr⁻¹) and phosphate fertilizers (0.4 Tg P yr⁻¹). The primary production of pasture incorporates 10.0 Tg P yr⁻¹ of P into grass biomass that is digested by animals, and the leaching and runoff loss averages 1.7 Tg P yr⁻¹. From all these fluxes, we estimated a global pasture PUE of 0.72, and a net accumulation of 2.2 Tg P yr⁻¹ in the soil. In the countries where grass P removal exceeded the labile P inputs, the labile soil P pool was depleted by 1.4 Tg P yr⁻¹. In the countries where the labile P input exceeded grass P removal, an average of 5.3 Tg P yr⁻¹ was transferred from the labile to the stable soil P pool from 2002 to



327 2010.

328 **3.1.5 Global P fluxes in livestock**

329 The annual P input to livestock was $25.6 \text{ Tg P yr}^{-1}$, with most of contributions from
330 grazed grass ($10.0 \text{ Tg P yr}^{-1}$) and processed feed ($10.0 \text{ Tg P yr}^{-1}$). The economic P output
331 in the form of livestock products averaged 1.5 Tg P yr^{-1} , which gives a PUE of 0.06.
332 Averages of 29% and 56% of the P produced in livestock manure were recycled into
333 cropland and pasture, respectively; the rest of this manure (4.3 Tg P yr^{-1}) was lost to the
334 environment.

335 **3.1.6 Global P fluxes in human use**

336 Humans receive an annual input of $14.0 \text{ Tg P yr}^{-1}$ from harvested crop products,
337 livestock products, and the use of detergents and other products manufactured from
338 phosphate rock. Although P inputs as food (crop food and livestock products) amounted
339 to 6.8 Tg P yr^{-1} , humans only absorbed 3.0 Tg P yr^{-1} (44%), the remainder being either
340 wasted before consumption (e.g., in food processing) or transferred back to livestock
341 as processed feed. Thus, only 14.3% of the total P inputs into the agriculture system
342 end up as food being actually consumed by humans. P lost to the environment by human
343 use amounts to 2.6 Tg P yr^{-1} , which is divided among 2.2 Tg P yr^{-1} lost through
344 inefficient processing and excreta and 0.4 Tg P yr^{-1} through bioenergy-related
345 emissions. The fate of non-recycled P in human waste was not separated between
346 bodies of water (untreated sewage) and landfill.

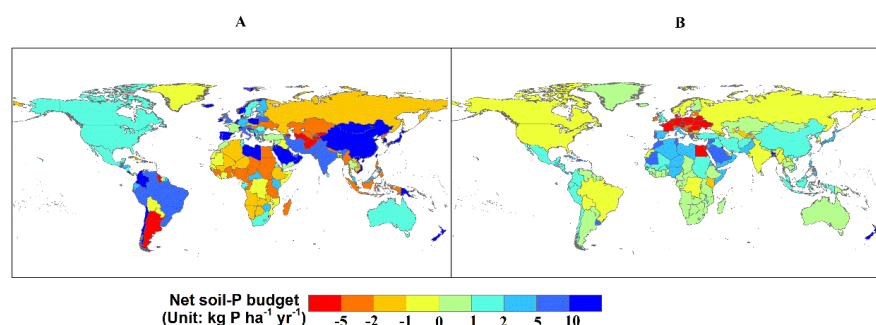
347 **3.2 Regional P budgets**

348 Cropland and pasture soils accumulated 59.6 and 19.4 Tg P from 2002 to 2010,
349 respectively. For croplands, the net P accumulation in the stable P pools amounted to
350 52.7 Tg P , and the remaining 6.9 Tg P accumulated in soil labile pools. For pasture, the
351 accumulation in the stable P pool was 25.0 Tg P , but 5.6 Tg P was transferred from the



352 stable P pool to be incorporated by grass in regions where P inputs are lower than grass

353 P uptake.



354

355 Figure 4: Map of global net soil P budgets (positive values, increase; negative values,
356 decrease) for (A) cropland and (B) pasture.

357 Those global numbers mask large regional differences (Table 1, Fig. 4A). About
358 32% of the global cropland area (in 75 countries) had annual soil P deficits from 2002
359 to 2007. This fraction increased to 50% in 2008 and 2009, at the time of the global
360 financial crisis, as a result of high P prices and the resulting reduction in fertilizer
361 application (Cordell *et al.* 2009, 2012), but returned close to the decadal mean value in
362 2010. On average, 48% of cropland P uptake was supplied by stable P that accumulated
363 in previous years according to the equations in section 3 of the Supporting Information.
364 Including the United States, France, Russia, Argentina, and Paraguay, 89 countries had
365 labile P inputs into cropland that were lower than crop P removal from 2002 to 2010.
366 However, if we consider stable P inputs, cropland soil still presented a net soil P surplus
367 in the United States during the same period. For pasture, a slightly smaller proportion
368 of the total global pasture area (43%) had a net annual soil P deficit from 2002 to 2010,
369 mostly in Europe and North America. However, only 48 countries had labile P inputs
370 into pasture that were lower than the P removal in grass.

371



3.2.1 Regional cropland budgets

Examining Figure 4A reveals that cropland in all African countries experienced an annual soil P deficit, especially in western and central Africa, with soil P loss rates per unit area ranging from 2.5 kg P ha⁻¹ yr⁻¹ in 2002 to 2.7 kg P ha⁻¹ yr⁻¹ in 2010. In contrast, cropland in Eastern Asia accumulated 23.4 kg P ha⁻¹ yr⁻¹ during the period from 2002 to 2010, a cumulative storage equivalent to more than four years of P fertilizer application. Cropland in Oceania, Europe, and the Caribbean and Central America also annually accumulated P in their soils. Cropland soils in North America and South America accumulated P from 2002 to 2007, but experienced temporary P deficits from 2008 to 2010. Yet despite this, crop yields did not decrease from 2008 to 2010 in those two regions, probably because of the re-mobilization of P that accumulated in stable pools. Cropland soils in western and central Asia were nearly balanced, with a mean areal flux of 0.2 kg P ha⁻¹ yr⁻¹.

Considering the different countries (Fig. 4A), the largest cumulative soil P increase was found in China (34.6 Tg P) for the 9 years from 2002 to 2010, followed by India (11.4 Tg P) and Brazil (3.6 Tg P). Pakistan (1.8 Tg P), the United States (1.8 Tg P), and New Zealand (1.8 Tg P) also had net soil P accumulation, yet of a smaller magnitude. These six countries accounted for 77% of the global accumulation of P in countries where cropland had a positive soil P balance. Furthermore, a large amount of P accumulated in the soil labile P pools of cropland in China and India, at about 20.0 and 4.5 Tg P, respectively; however, in the United States, about 6.0 Tg P accumulated in the cropland stable P pool from 2002 to 2010; thus, 4.2 Tg P was absorbed from the previous cropland soil P. In contrast, most African countries experienced persistent cropland soil P deficits from 2002 to 2010. This was especially true in Nigeria, which had a cumulative deficit of 1.7 Tg P (Fig. 4A). We also found cumulative soil P deficits



397 in Russia, the Ukraine, and Kazakhstan, but with a smaller magnitude (1.1, 0.9, and 0.7
398 Tg P, respectively) for the 9 years. Comparing the rates of change of crop soil P per
399 unit area, New Zealand had the fastest rate of increase ($>100 \text{ kg P ha}^{-1} \text{ yr}^{-1}$), whereas
400 Argentina had the fastest rate of decrease ($-7.9 \text{ kg P ha}^{-1} \text{ yr}^{-1}$). In terms of the difference
401 between inputs and outputs, loss rates in Argentina were about five times input rates.

402 3.2.2 Regional pasture budgets

403 We found mainly net losses of P in pasture soils (Fig. 4B), most likely because of
404 the net removal of P through animal grazing followed by the export of manure P to
405 enrich cropland soils. Pasture soil P loss rates per unit area in Europe averaged 0.4 kg
406 $\text{P ha}^{-1} \text{ yr}^{-1}$ and reached high values in countries (Denmark, Luxembourg, Germany, and
407 Belgium) with intensive livestock production systems (Chang *et al.*, 2015) and large
408 grass consumption by livestock, with loss rates $>10 \text{ kg P ha}^{-1} \text{ yr}^{-1}$). North American
409 pastures had a smaller average loss rate of about $0.1 \text{ kg P ha}^{-1} \text{ yr}^{-1}$. The United States,
410 India, and Russia had the largest cumulative P deficits, at 2.1, 1.5, and 0.7 Tg P,
411 respectively, from 2002 to 2010. In contrast, pasture in the Caribbean and Central
412 America had greater P inputs than P removals. Consequently, these regions had the
413 largest soil P accumulation rates. Pasture in Northern and Eastern Africa also had net
414 soil P accumulation. For instance, Mauritania, Tunisia, and Morocco had net soil P
415 accumulation rates of 9.8, 9.4, and $5.5 \text{ kg P ha}^{-1} \text{ yr}^{-1}$, respectively. The reason for this
416 excess is not clear, but one possibility is that these countries apply P fertilizer to some
417 of their pasture.



418

Table 1: Regional annual agricultural P budgets and P-use efficiency (PUE)

Subsystem	World	Eastern and Southern Africa	Northern Africa	Western Central Africa	Eastern Asia	Southern and Southeastern Asia	Western Central Asia	Oceania	Europe	North America	Caribbean and Central America	South America
Cropland	6.6	-0.1	-0.1	-0.2	4.1	1.4	0.0	0.3	0.7	0.3	0.0	0.3
Pasture	2.2	0.1	0.5	0.3	0.5	0.2	0.2	0.4	-0.3	-0.1	0.1	0.3
					Agricultural land P budget per unit area (kg P ha ⁻¹ yr ⁻¹) ¹							
Cropland	4.7	-1.0	-1.5	-2.7	23.4	4.1	0.2	5.2	2.8	1.5	3.8	2.3
Pasture	0.4	0.2	1.6	0.9	1.0	0.4	0.6	0.8	-0.4	-0.1	3.4	0.4
					Food consumption (Tg P yr ⁻¹)							
Crops	5.3	0.2	0.3	0.2	1.4	1.3	0.3	0.0	0.8	0.4	0.0	0.3
Meat	0.71	0.01	0.01	0.01	0.29	0.05	0.01	0.01	0.17	0.10	0.00	0.05
Eggs	0.16	0.00	0.00	0.00	0.07	0.02	0.01	0.00	0.03	0.02	0.00	0.01
Milk	0.60	0.01	0.03	0.00	0.04	0.13	0.04	0.01	0.18	0.09	0.01	0.05
					PUE							
Cropland	0.46	0.80	0.84	1.51	0.27	0.43	0.64	0.31	0.54	0.57	0.53	0.63
Pasture	0.72	0.77	0.61	0.46	0.58	0.80	0.61	0.42	1.25	0.98	0.37	0.75
Livestock	0.06	0.02	0.02	0.01	0.08	0.05	0.04	0.04	0.09	0.08	0.03	0.03
Food	0.45	0.60	0.50	0.64	0.40	0.64	0.44	0.26	0.28	0.32	0.68	0.42
					International trade of P in commodities (Tg P yr ⁻¹) ²							
Crops	-	0.02	0.12	0.03	0.35	0.03	0.12	-0.07	-0.02	-0.41	0.03	-0.24
Meat	-	0.000	0.001	0.000	0.013	0.000	0.002	-0.004	-0.002	-0.005	0.001	-0.010
Eggs	-	0.0001	0.0000	0.0000	0.0001	-0.0004	0.0001	0.0000	0.0000	-0.0002	0.0000	-0.0001
Milk	-	0.000	0.003	0.001	0.004	0.005	0.004	-0.015	-0.012	0.003	0.001	-0.001
Fertilizer	-	0.05	-0.67	0.03	0.00	1.45	-0.05	0.21	-0.37	-1.00	0.06	1.10

¹ The positive values represent a soil P surplus, whereas negative values represent a soil P deficit.

² The positive values represent net P importers, whereas negative values represent net P exporters.

419

420



3.3 Phosphorus-use efficiencies in different regions

Table 1 gives the values of PUE for cropland, pasture, livestock, and food (human use) in the world's different regions. Globally, 116 countries have cropland PUE values above the global mean value of 0.46, mostly in Africa, and these countries account for 64% of the global cropland area. In addition, 16% of the countries had a PUE of around 0.6 (0.55 to 0.65). In particular, African countries had the highest overall cropland PUE (≥ 0.80) because of their low P input. On the other hand, Eastern Asia and Oceania have cropland PUE below the global average. Conversely, pasture had high PUE in Europe (1.25) and North America (0.98) but low values in Africa (≤ 0.77) and particularly low values in the Caribbean and Central America (0.37). P removal from pasture exceeded P inputs in Europe, resulting in pasture PUE > 1 , largely because of P inputs from feed given to animals.

The livestock subsystem generally had a low PUE (< 0.1), with the highest values in Europe, North America, and Eastern Asia (Table 1). Regarding human food PUE, our data indicate that only 25% to 40% of the P in food products in Eastern Asia, Oceania, Europe, and North America is actually consumed by humans (Table 1). The resulting low PUE of human use in these regions results from both large P inputs and high food waste. Eastern and Southern Africa, Western and Central Africa, Southern and Southeastern Asia, and the Caribbean and Central America had the highest PUE for human use, with more than 60% of P in food being consumed by humans. Globally, most of the P consumed by humans (78%) originates from crops, and the fraction of P from livestock differs among regions; it ranges from 35% of the total human food P consumption in Oceania, Europe, and North America to 10% in less developed regions (Africa and the Caribbean and Central America) and to 4% in Western and Central Africa.



3.4 P Flows through international trade

Approximately 2.1 Tg P yr⁻¹ entered into international trade in 2010, amounting to about 17% of the total harvested crop P (Figure 5). The remainder (10.6 Tg P yr⁻¹) is consumed domestically. Differences in crop types as a result of their specific P content (Table SI-1) strongly determine the magnitude of the traded P fluxes. For example, 37% of the P in soybean and 27% of the P in wheat produced each year were traded internationally in 2010. Also significant fractions of the P in maize, other cereals, and fruit were traded internationally, but almost all of the P in sugar crops and fiber were consumed or processed in the countries where they were grown.

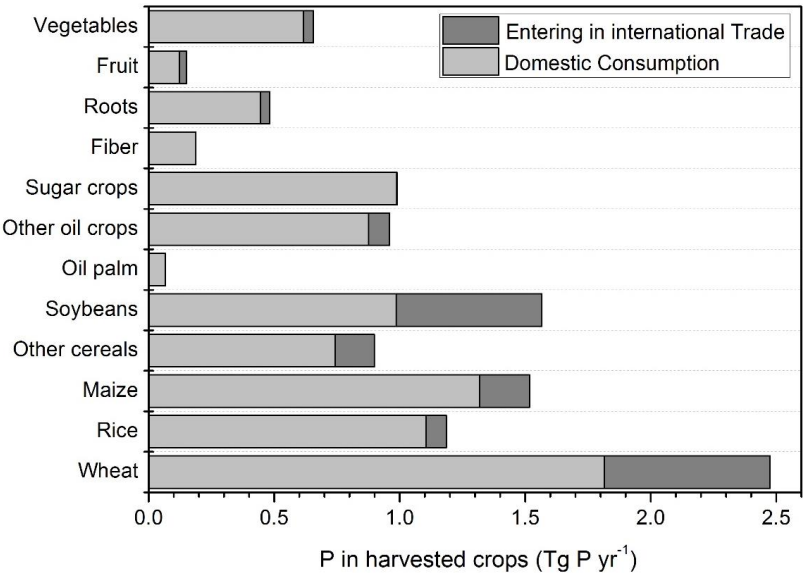


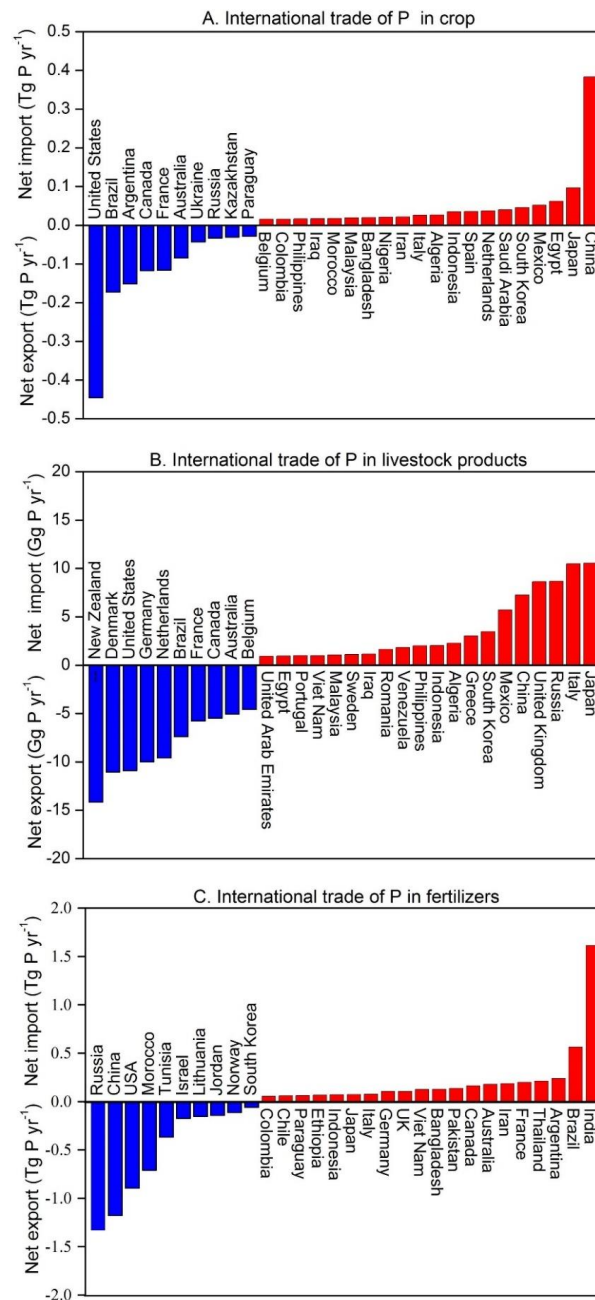
Figure 5: P flows embedded in different crop products, including the fraction of these flows entering into international trade circuits vs. being used for domestic consumption for the year 2010.



461 Considering the P fluxes in phosphate fertilizers and food products, we examined
462 how international trade influences regional P budgets and redistributes P between
463 regions. We found that Southern and Southeastern Asia have the largest net P imports
464 (Table 1), with imports of phosphate fertilizer amounting to 1.4 Tg P yr⁻¹ and P exports
465 as food products being much smaller, mainly to China and South Korea. South America
466 is the second-largest exporter of P in food, but imports 56% of its P fertilizer. North
467 America is a large exporter of P in both crop products and fertilizer, yet it also imports
468 P-rich milk products. Most European countries imported nearly all their phosphate
469 fertilizers, but Europe as a whole is a net exporter because of large exports (0.9 Tg P
470 yr⁻¹) from Russia (Figure 6). Western European countries were the main exporters of
471 P-rich livestock products. Some Northern African countries (especially Morocco and
472 Tunisia, which have the largest mines of P-rich ores), exported a total of 0.7 Tg P yr⁻¹
473 in fertilizer. The remaining regions (Eastern and Southern Africa, Northern Africa, and
474 the Caribbean and Central America) imported P in both food and fertilizer, although
475 much less than other regions (Table 1).

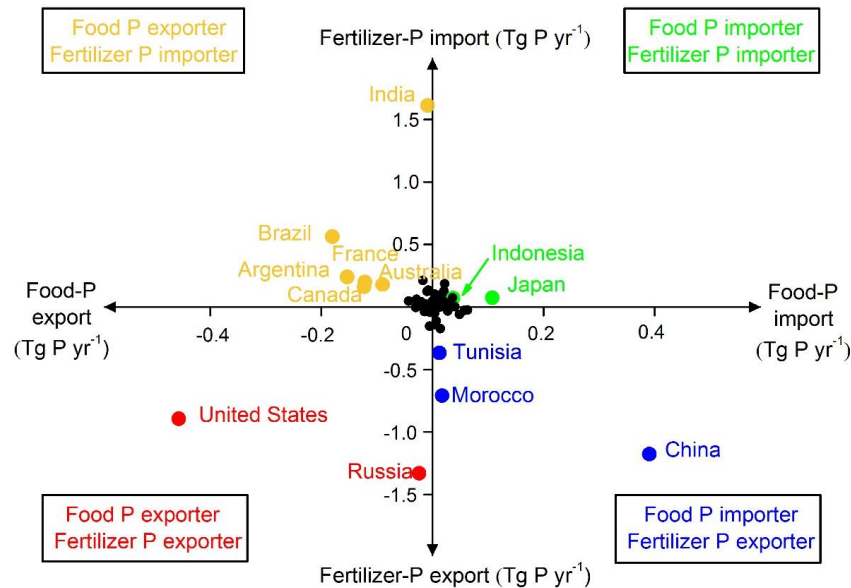
476 Figure 6 illustrates the disparities among countries with respect to the role of
477 international trade in crops, livestock, and fertilizer for the main exporters and
478 importers. Based on data for all 224 countries, a country can be categorized into one of
479 the following four groups (Figure 7):

480



481

482 Figure 6: Annual P flows embedded in traded crop products (A), livestock products (B),
 483 and fertilizers (C) in 2010. By convention, a positive flow is P received (imported) by
 484 a country.



485

486 Figure 7: Groupings of the countries based on whether they import or export P through
 487 their international trade in food and fertilizer.

488 Food and fertilizer P exporters: P storage in these countries has been decreasing
 489 due to their international exports of both fertilizer and food. Examples include the
 490 United States and Russia.

491 Food P importers and fertilizer P exporters: This group mainly comprises countries
 492 that export phosphate fertilizers and import food to meet domestic consumption.
 493 Examples includes Tunisia, Morocco, and China.

494 Food P exporters and fertilizer P importers: These countries have high food and
 495 livestock production, but this depends strongly on phosphate fertilizer imported from
 496 other countries. Examples include Brazil, Argentina, Canada, France, Australia, and
 497 India.

498 Food and fertilizer P importers: These countries depend on imports for both food
 499 and fertilizers; they are thus vulnerable to economic shocks that result from changing
 500 food prices. Examples include Japan and Indonesia.



501 International trade affects the global P cycle by physically moving the P contained
502 in traded crops, livestock products, and phosphate fertilizers (Grote *et al.*, 2005).
503 Imports of P fertilizers accounted for 55% and 79%, respectively, of the total
504 application of P fertilizer for countries that are food P exporters and fertilizer P
505 importers or food and fertilizer P importers. The P trade in food followed a similar trend.
506 Countries that are food P importers and fertilizer P exporters or food and fertilizer P
507 importers depended more on food imports than countries that are food and fertilizer P
508 exporters or food P exporters and fertilizer P importers. International trade also
509 increased the connections among countries (Table 2). For example, although the United
510 States and China are clearly major P fertilizer exporters, they also import fertilizer from
511 each other; 2.6% of the P fertilizer applied in the United States originated in China, and
512 3.6% of the phosphate fertilizer applied in China originated in the United States. In
513 addition, 11.4% of the phosphate fertilizer consumption in the United States originated
514 from Russia, Morocco, Tunisia, and other countries. About 1.5% of Chinese domestic
515 P consumption originates from the United States, which is higher than the fraction of
516 domestic P consumption in the United States from China. Countries with no or small
517 reserves of P-containing minerals imported large amounts of phosphate fertilizer; for
518 example, imports accounted for 61 and 46% of total P consumed in France and Brazil
519 (food P exporters and fertilizer P importers), and 76% of total P consumed in Japan.
520



Table 2: Proportions of total consumption and total international trade accounted for by P in fertilizer and food imports and exports.

Proportion (%)				
Group	P fertilizer imports as a proportion of total consumption	P fertilizer exports as a proportion of the total international P fertilizer trade	P in food imports as a proportion of total consumption	P in food exports as a proportion of the total international P in the food trade
Group Level				
Food and fertilizer exporter	22	43	7	31
Food importer and fertilizer exporter	5	48	22	5
Food exporter and fertilizer importer	55	5	5	48
Food and fertilizer importer	79	4	28	15
Country level				
United States (food and fertilizer exporter)	13	18	6	26
China (food importer and fertilizer exporter)	2	20	14	2
France (food exporter and fertilizer importer)	52	0	19	8
Brazil (food exporter and fertilizer importer)	44	1	4	10
Japan (food and fertilizer importer)	40	0	60	0

3.5 Uncertainties in soil P changes result from uncertain P concentrations

We estimated the net cropland soil P balance in 2000 by means of Monte Carlo simulations, as described in section 2.7. We found a net accumulation of 5.8 ± 0.6 Tg P yr⁻¹. More detailed calculations suggest that uncertainty in the crop P concentrations contributed ± 0.2 Tg P yr⁻¹ of the uncertainty in the net cropland soil P balance; this is because of dominance of the calculations by cereals, which have low uncertainty due



530 to the narrow range of reported P concentrations (Antikainen *et al.*, 2005; COMIFER,
531 2007; USDA-NRCS, 2009; Waller, 2010). Uncertainty in P concentrations in crop
532 residues contributed an additional $\pm 0.2 \text{ Tg P yr}^{-1}$ to the total uncertainty, and uncertainty
533 in P concentrations in the livestock manure applied to cropland added $\pm 0.4 \text{ Tg P yr}^{-1}$.
534 In addition, the uncertainty in the pasture soil P balance attributed to uncertainty in the
535 P concentrations in grass biomass and manure was $\pm 1.3 \text{ Tg P yr}^{-1}$. This relative
536 uncertainty is higher than that for the cropland soil P balance, and this results from the
537 large range of grass P concentrations found in our review of the available data. See
538 Table SI-5 for more details.

539 **4. Discussion**

540 **4.1 Cropland PUE and P in harvested crops as a function of cropland P inputs**

541 Figure 8A shows the relationship between the cropland PUE and cropland P inputs
542 for 35 countries that are large crop producers. PUE decreased exponentially with
543 increasing input; that is, P was used most efficiently at low application rates. PUE
544 decreased rapidly as P inputs increased to $10 \text{ kg P ha}^{-1} \text{ yr}^{-1}$, and then decreased more
545 slowly. High PUE values were associated with countries that had a low P input and a
546 soil P deficit. This suggests that there is a trade-off between efficient use of P in
547 cropland and the avoidance of soil P deficits that limit crop yields (Obersteiner *et al.*,
548 2013). Figure 8A also indicates that cropland soils have a net soil P deficit if their inputs
549 are lower than $10 \text{ kg P ha}^{-1} \text{ yr}^{-1}$, which is a threshold value that corresponds to $\text{PUE} =$
550 0.67. Argentina, South Africa, Indonesia, Mexico, and Paraguay are below this
551 threshold (Figure 9).

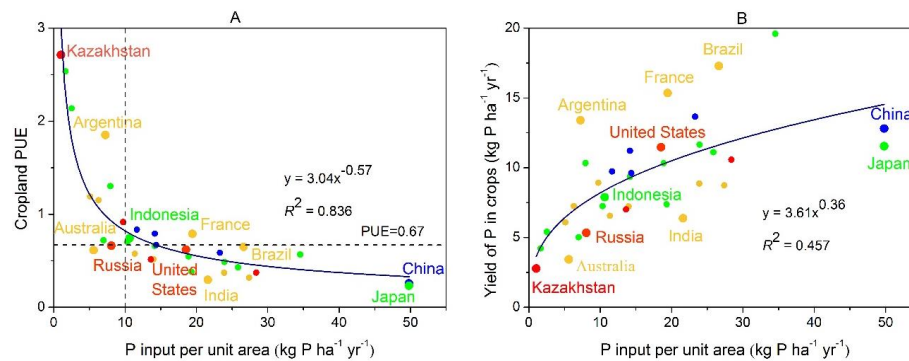


Figure 8: The relationships between P input per unit area of cropland and (A) phosphorus-use efficiency (PUE) The horizontal line at PUE = 0.67 represents the global average. (B) P in harvested crops for the 35 largest crop producers representing 90% of global crop. The equations give the fit to the data represented by black curves.

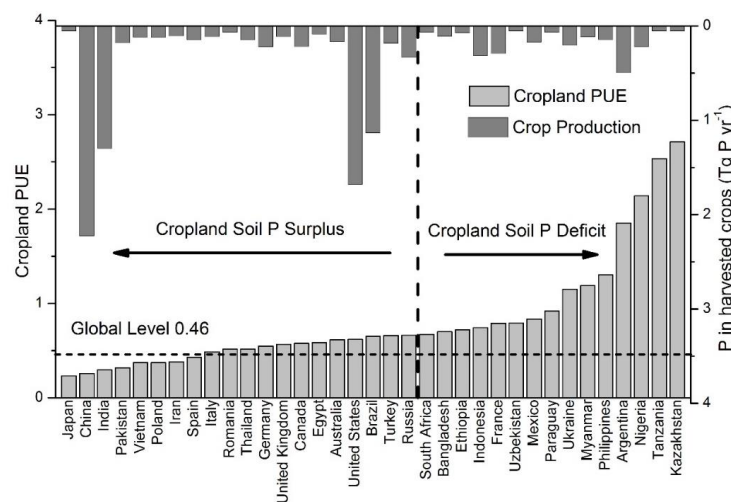


Figure 9: Phosphorus-use efficiency (PUE) and P in harvested crops for the 35 large countries shown in Fig 8. Cropland soil P surplus or deficit is separated by the vertical dashed line

P in harvested crops increased exponentially with increasing P inputs, but the response slowed at high P inputs (Fig. 8B). The P in harvested crops in countries with



563 cropland PUE > 0.67 (except Argentina) is only half of that in countries with high P in
564 the harvested crops, such as the United States and China. P in the harvested crops was
565 very low in Australia due to low cropland P input, which was less than 25% of the
566 inputs in the United States and China. P already present in the soil may be sufficient to
567 sustain high crop yields for some time without additional inputs in some countries (e.g.,
568 France) that formerly had large P fertilization rates, despite currently having a negative
569 annual P balance. Comparing Figures 8A and 8B suggests that total cropland P inputs
570 of 20 to 25 kg P ha⁻¹ yr⁻¹ may be a good compromise that will achieve high yields while
571 creating a near-equilibrium soil P balance. Both excessive P inputs (e.g., China and
572 Japan) and low PUE (e.g., India) can lead to high P accumulation in cropland soil,
573 leading to high losses into the environment.

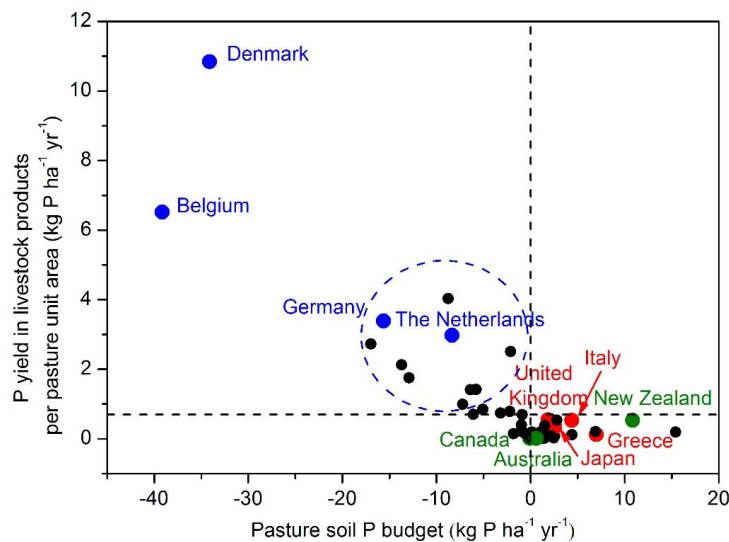
574 The data in Figure 8 indicate that different countries face different challenges for
575 P resource management, implying a need for country-specific policy options and
576 solutions. Countries like Kazakhstan and Argentina may have to increase P inputs to
577 their cropland in order to prevent long-term depletion of soil P, which could be realized
578 by increasing the application of phosphate fertilizer or reducing losses to leaching and
579 erosion. Countries like France that are currently experiencing a net negative soil P
580 balance (Fig. 9) following a period of sustained accumulation (Senthilkumar et al. 2012;
581 van Dijk *et al.* 2016) may need to progressively adjust fertilizer inputs in coming years
582 to balance inputs with removals and avoid the risk of a long-term soil fertility decline
583 due to inadequate levels of P. In contrast, countries such as Japan and China are rapidly
584 accumulating P in cropland soils due high and sustained P inputs, and will urgently
585 need to consider how to improve their cropland PUE. This could be initiated by
586 identifying crop types that are being over-fertilized and regions with excessive
587 application of phosphate fertilizer; they can then consider a range of options such as



588 precision agriculture (i.e., applying only as much P as the crop requires). We estimate
589 that if Chinese cropland PUE could be increased to the global average of 0.46 (Fig. 9),
590 China would save 3.8 Tg P yr⁻¹ of phosphate fertilizer, which is equivalent to 60% of
591 its phosphate fertilizer consumption in 2010. Last, in countries like India where crop P
592 harvests are lower than average despite high average P inputs and positive soil ΔP ,
593 improvements in agricultural management (such as the use of precision fertilization)
594 appear necessary. We did not have access to sub-national data for this study, but it is
595 likely that in a country as large as India, some regions, crop types, or region–crop type
596 combinations may have excessive or insufficient P input.

597 **4.2 Pasture P budget, livestock consumption, and international trade**

598 Figure 10 shows that the soil P balance is negatively related to the flux of P in
599 livestock products per unit area of pasture. Several western European countries
600 (Germany, the Netherlands, Denmark, and Belgium) achieve high P yields in livestock
601 products (defined by the amount of P in livestock products per unit area of pasture),
602 and all of these countries export livestock products. In these countries, only a small
603 fraction of livestock manure is recycled to pasture, so there is currently a soil P deficit;
604 in the long term, this may result in a loss of soil fertility. Therefore, these countries
605 should increase P fertilization in pasture or import forage or feed to supply the P
606 required to sustain high livestock production. New Zealand, Australia, and Canada are
607 also large exporters of P in livestock products. However, given their low-input
608 production systems and large areas of pasture (Fig. 4B), P removals per unit area
609 through grazing are much lower than in Western Europe, and the soil P balance of
610 pasture ranges from slightly negative to slightly positive.



611

612 Figure 10: The relationship between the P yield of livestock products, defined by the
 613 amount of P in livestock products per unit area of pasture and the P balance of pasture
 614 soils.

615 4.3 Livestock and human food PUE, and trends in P consumption

616 Increasing consumption of livestock products by humans is an essential factor that
 617 is responsible for increasing P mining and increasing P inputs to agricultural systems
 618 (Metson *et al.*, 2012; van Dijk *et al.*, 2016). Where socioeconomic development is
 619 improving the income of residents, especially in Africa and the Caribbean and Central
 620 America region, residents are consuming more P from livestock products (Fig. SI-3).
 621 Unfortunately, the livestock PUE in countries in these two regions is much smaller
 622 (0.01 to 0.03) than the global average of 0.06 (Table 1), indicating that only a small
 623 proportion of livestock P inputs is used by humans. This may be because countries in
 624 these regions are primarily importers of livestock products. Therefore, animal
 625 husbandry has important implications for global P security and special attention will be



626 required to improve livestock PUE (Wu *et al.*, 2014). If livestock PUE reaches the
627 global level of 0.06 in these two regions, both regions could more than double their
628 livestock production, by about 0.16 Tg P yr⁻¹.

629 In addition, the management of manure differs greatly among regions due to their
630 different livestock production systems. The yield of livestock products is very low in
631 African countries, resulting in low livestock PUE. Almost all livestock manure is
632 applied to cropland, where this resource is an important P input. In contrast, with
633 application of phosphate fertilizer to pasture in Europe and Eastern Asia, only a small
634 fraction of livestock manure is recycled for pasture (36 and 17%, respectively); a larger
635 fraction of the manure is applied to cropland in Eastern Asia (40%) and Europe (60%).
636 Consequently, improving the manure utilization efficiency and applying more livestock
637 manure to pasture will be important strategies in Eastern Asia and Europe (Wu *et al.*,
638 2014).

639 As shown in Section 3.3, only 45% of the P that enters the food production
640 subsystem was absorbed by humans; thus, large amounts of food (and the P it contains)
641 are wasted, although some parts of the wastes were consumed by livestock. Despite this
642 recycling, 2.2 Tg P yr⁻¹ flowed into the environment as wastes either before or after
643 food consumption, and only 14.3% of the total P inputs to the agriculture system ended
644 up in food consumed by humans. In Eastern Asia, Oceania, Europe, and North America,
645 the PUE of human food was very low, reflecting the high proportion of livestock
646 products in the diet and a high degree of waste. Therefore, decreasing food waste before
647 consumption, recycling P in food waste, and better treatment of organic wastes could
648 significantly decrease the amount of P required to support humans (Metson *et al.*, 2012;
649 van Dijk *et al.*, 2016). In Eastern Asia, Oceania, Europe, and North America, fully
650 absorbing the 45% of the P that enters food produced for humans could reduce



651 agricultural inputs of P by 0.7 Tg P yr⁻¹ globally. Thus, decreasing food waste and
652 improving the PUE of human food represent key challenges that must be solved to
653 achieve sustainable P management.

654 Population increases and dietary changes are requiring higher P inputs in cultivated
655 land and increased mining of P ores (Grote *et al.*, 2005; Foley *et al.*, 2011). From 2002
656 to 2010, this mining increased by 33% in our estimate, during a period when the global
657 population and per capita food P consumption increased by 10 and 5%, respectively. In
658 2010, humans consumed 8.0% and 3.8% more P in livestock products and crops,
659 respectively. Since livestock PUE was much lower than cropland PUE, consumption of
660 more livestock products resulted in lower external P inputs in food that flowed into the
661 human subsystem; this proportion decreased from 36% in 2002 to 31% in 2010.
662 Therefore, consuming more livestock products will require increasing P inputs. Thus,
663 human dietary shifts may have been responsible for half of the increase of P ore mining.

664 **4.4 International trade and global P flows**

665 International trade also increased the connections among countries. Whether
666 international trade is good or bad for humans and the environment in terms of its impact
667 on the management of P resources is a complex question. International trade can
668 increase cropland P deficits if countries that export large amounts of P in crop and
669 livestock products do not counteract these exports by increasing inputs of phosphate
670 fertilizer to soils. For example, Argentina exported lots of food to other countries (about
671 0.15 Tg P yr⁻¹), and has developed a serious cropland soil P deficit of 0.38 Tg P yr⁻¹
672 (10.3 kg P ha⁻¹ yr⁻¹). Massive P imports through trade can result in an excess supply of
673 P to cropland soils as manure (Schipanski and Bennett, 2012), with potentially
674 significant negative environmental effects. On the one hand, trade can hamper the
675 proper recycling of P resources from wastes and manure to agricultural soils through



676 local food webs (Schipanski and Bennett, 2012). On the other hand, trade may
677 contribute to more efficient use of P resources if traded products flow from countries
678 with lower PUE to countries with higher PUE, as is generally observed for water
679 resources (Dalin et al., 2014). This confirms that more integrated studies are required
680 to fully assess the effects of trade on P resource recycling, efficiency, and conservation.
681 Our study identified world regions and countries with lower PUE and others with high
682 PUE, and regions and countries with net loss of P in soils and others with net gain. This
683 provides valuable information to policymakers on how to improve the trade
684 relationships for a global optimization of PUE and therefore global food security.

685 **4.5 Comparison with previous studies**

686 Previous studies have estimated P flows in agriculture at a global scale (Smil, 2000;
687 Sheldrick *et al.*, 2003; Liu *et al.*, 2008; Cordell *et al.*, 2009; Bouwman *et al.*, 2009,
688 2013; Potter *et al.*, 2010; MacDonald *et al.*, 2011). However, to the best of our
689 knowledge, the present analysis provides the first consistent multi-year overview of the
690 P flows in agriculture. In addition, it provides national and regional P budgets,
691 calculates agricultural PUE, and quantifies P fluxes in international trade based on a
692 combination of datasets for cropland and pasture inputs (fertilizers, manure,
693 atmospheric deposition, and recycling of crop residues) and outputs (crop harvests,
694 residue removal, and P loss by burning and leaching or surface runoff into bodies of
695 water). For data from 2000, our results are consistent with the abovementioned studies
696 for most P flows (Table 3). For data from 2000, our results are generally consistent with
697 those in the previous studies for cropland soil P inputs, harvested crop P, cropland soil
698 P lost by erosion or surface runoff into bodies of water, pasture soil P inputs, and
699 harvested grass P (Table 3). However, methods, data sources, and system boundaries
700 differed among the studies, making an accurate comparison difficult. Our estimate of a



net accumulation of 5.8 ± 0.6 Tg P yr⁻¹ is in line with the reported net accumulation in soils, which ranged between 0 and 11.5 Tg P yr⁻¹ (Smil, 2000; Bennett *et al.*, 2001; Bouwman *et al.*, 2009; MacDonald *et al.*, 2011), but disagrees with the estimate of Liu *et al.* (2008), who calculated a net loss of 9.6 Tg P yr⁻¹. The difference from the present results can be explained by accounting for large P losses (19.3 Tg P yr⁻¹) due to soil erosion caused by land use change and over-grazing. The quantification of erosional losses of P from arable land is prone to high uncertainties due to the unknown amount of redeposited soil material, and other studies have reported much lower losses (e.g., 2.5 Tg P yr⁻¹; Quinton *et al.*, 2010).

Table 3: Comparison of the present results for P flows and budgets in 2000 with results of other studies at a global level (Tg P yr⁻¹).

	Global P flux	Previous studies	Our study	Reasons for differences
	Fertilizer input	14–15 ^{1–3}	13.7	–
	Animal manure to cropland	6–8 ^{2,3}	6.7 ± 0.4	Method
	Human sewage sludge to cropland	1.5 ^{1,3}	1.3	Method
	Crop production	8.2–12.3 ^{1–5}	10.2 ± 0.4	Boundary/Data
Cropland	Crops (human food)	3.5 ³	4.8 ± 0.2	Method/Data
	Crops (animal feed)	2.6 ³	1.9 ± 0.1	Data
	Crop residues	3.75–4.5 ^{1–2}	6.7 ± 0.2	Method/Data
	Recycling of residues	1–2.2 ^{1–3}	3.5 ± 0.1	Method/Data
	Leaching and runoff from cropland	4 ⁶	3.2	Method
	Livestock manure	17.1–24.3 ^{5,7,8}	22.3 ± 1.3	Method/Data
	Manure wasted (released into the environment)	2–8 ^{1–3}	4.1 ± 0.2	Method/Data
Pasture	Grass	6–12.1 ^{3,4}	8.9 ± 1.3	Method/Data
	Animal feed additives	0.9 ³	1.4	Data
	Leaching and runoff from pasture	1.0 ⁵	1.6	Method
Humans	Excreta	3–3.3 ^{1,3}	2.8	Method

Sources: 1. Liu *et al.*, 2008; 2. Smil, 2000; 3. Cordell *et al.*, 2009; 4. MacDonald *et al.*, 2011; 5. Bouwman *et al.*, 2009; 6. Bouwman *et al.*, 2011; 7. Sheldrick *et al.*, 2003; 8. Potter *et al.*, 2010.

The main cropland P fluxes estimated in our study agreed with previous results, except for the production and recycling of crop residues (Table 3). Smil (2000) and Liu *et al.* (2008) used harvest index data (defined as the ratio of total aboveground biomass



718 to crop residues) for estimating the P in crop residues, whereas we estimated P in crop
719 residues by combining data from Liu *et al.* (2008) and FAO. MacDonald *et al.* (2011)
720 estimated that 29% of the global cropland area was subject to soil P deficits in 2000,
721 which is similar to our estimate (32%) based on data from 2002 to 2010. In addition,
722 our estimate of 22.3 Tg P yr⁻¹ in animal manure for the livestock subsystem in 2000 is
723 within the reported range of 17.1 to 24.3 Tg P yr⁻¹ from Potter *et al.* (2010). We defined
724 global cropland PUE as the ratio of P in harvested crops to total P inputs, without
725 accounting for recycling of crop residues. Under this definition, global PUE was
726 estimated to be 0.43 by Liu *et al.* (2008) and 0.40 by Smil (2000), both of which are
727 comparable to our estimate of 0.46 from 2002 to 2010. Since we applied the same
728 methods across the globe to calculate agricultural P fluxes, we were able to compare
729 the P fluxes and budgets for different regions and countries on a consistent basis. This
730 information is of critical importance for the development of more appropriate
731 agricultural policy and to support the development of technological and other solutions
732 for different types of countries, which better integrate cultivated ecosystems, livestock
733 production, and the human food supply.

734 **4.6 Limitations and novelty of our study**

735 Due to limited data sources for some parameters, our study and most previous
736 studies focused on P in livestock products and manure as the outputs of the livestock
737 system, and did not consider the fate of P in non-edible livestock products (e.g., bones,
738 blood, leather products). Xu *et al.* (2005) pointed out that from 12 to 23% and 72% of
739 P were contained in livestock meat and bones, respectively. If these percentages are
740 applied to our data, this gives an annual flux of 2.5 Tg P yr⁻¹ in the bones of slaughtered
741 animals. Although most livestock bones are currently wasted or landfilled, some
742 countries have begun to use them as fertilizers, protein sources, and condiments (Wu



743 and Ma, 2005; Li, 2008). In addition, as we focused on the annual P budgets for
744 livestock and human beings, we did not account for P accumulation in humans. From
745 2002 to 2010, the global population increased by 635×10^6 persons. If we assume that a
746 typical adult body contains 600 g of P, then about 0.38 Tg more P would have
747 accumulated in humans. Therefore, the annual human P accumulation would be 0.04
748 Tg P yr⁻¹, accounting for only 0.3% of the P inputs into humans.

749 Despite the abovementioned limitations in our study, we were able to achieve some
750 interesting and novel results. First, we have provided a detailed and harmonized
751 summary of the P fluxes as inputs and outputs for the agricultural system and the
752 internal P flows within the agricultural system at national, regional, and global scales.
753 In addition, we have characterized the P budgets and P-use efficiencies in the
754 subsystems of the overall agricultural system, and have discussed their influences and
755 impacts. Finally, we have discussed how changes in population, diets, and food
756 consumption have influenced global mining of P ore and how international trade has
757 influenced P fluxes. These insights will support the development of policies to use P
758 more sustainably at national, regional, and global levels.

759 **Data availability**

760 The global and regional phosphorus budgets and their PUEs in agricultural systems
761 is publicly available at <https://doi.pangaea.de/10.1594/PANGAEA.875296>.

762 **Conclusion**

763 The estimation of global and regional phosphorus budgets in agricultural systems,
764 as well as their PUE, is a major effort by anthropogenic nutrient cycle research
765 community that requires lots of work. We quantified in detail the P inputs and outputs
766 of cropland and pasture, and the P fluxes through human and livestock consumers of
767 agricultural products, at global, regional, and national scales from 2002 to 2010. The



768 results reveal the significant and imbalanced P budgets in cropland and pasture. The
769 hot spots of cropland P budgets shifted from increasing P accumulation in Eastern Asia
770 countries to increasing soil P deficits in African countries, while European and North
771 American pasture had a soil P deficit. There presents great differences among the values
772 of PUE for or cropland, pasture, livestock, and food at global, regional, and national
773 scales. PUE decreased exponentially with increasing input; that is, P was used most
774 efficiently at low application rates; meanwhile, P in harvested crops increased
775 exponentially with increasing P inputs, but the response slowed at high P inputs.
776 International trade played a significant role in P redistribution among countries through
777 the flows of P in fertilizer and food among countries. It can mitigate regional P
778 imbalances in agricultural soils, by optimizing phosphate fertilizer application and
779 recycling P.

780

781 Acknowledgments

782 This study was supported by the National Natural Science Foundation of China
783 (41571022, 41625001), the Beijing Natural Science Foundation (Grant 8151002), the
784 National Science and Technology Major Project (2015ZX07203-005), and a Synergy
785 Grant (ERC-2013-SyG-610028 IMBALANCE-P) from the European Research
786 Council.

787

788 References

789 Antikainen, R., Lemola, R., Nousiainen, J. I., Sokka, L., Esala, M., Huhtanen, P., and
790 Rekolainen, S. Stocks and flows of nitrogen and phosphorus in the Finnish food
791 production and consumption system. *Agr. Ecosyst. Environ.*, 107(2), 287-305,
792 2005.



- 793 ASAE. Manure production and characteristics. D384.2, American Society of
794 Agricultural Engineers, St. Joseph, MI, USA, 2005.
- 795 Bennett, E. M., Carpenter, S. R., and Caraco, N. F. Human Impact on Erodeable
796 Phosphorus and Eutrophication: A Global Perspective: Increasing accumulation
797 of phosphorus in soil threatens rivers, lakes, and coastal oceans with
798 eutrophication. *BioScience*, 51(3), 227-234, 2001.
- 799 Bouwman, A. F., Beusen, A. H., and Billen, G. Human alteration of the global nitrogen
800 and phosphorus soil balances for the period 1970–2050. *Global Biogeochem. Cy.*,
801 23(4), 2009.
- 802 Bouwman, L., Goldewijk, K. K., Van Der Hoek, K. W., Beusen, A. H., Van Vuuren,
803 D. P., Willems, J., Rufino, M. C., and Stehfest, E. Exploring global changes in
804 nitrogen and phosphorus cycles in agriculture induced by livestock production
805 over the 1900–2050 period. *PNAS*, 110(52), 20882-20887, 2013.
- 806 Carpenter, S. R., Bennett, E. M. Reconsideration of the planetary boundary for
807 phosphorus. *Environ. Res. Lett.*, 6, 014009, 2011.
- 808 Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., and
809 Smith, V. H. Nonpoint pollution of surface waters with phosphorus and nitrogen.
810 *Ecol. Appl.*, 8(3), 559-568, 1998.
- 811 Chang, J. F., Viovy, N., Vuichard, N., Ciais, P., Wang, T., Cozic, A., Lardy, R., Graux,
812 A. I., Klumpp K., Martin, R., and Soussana, J. F. Incorporating grassland
813 management in ORCHIDEE: model description and evaluation at 11 eddy-
814 covariance sites in Europe. *Geosci. Model Dev.*, 6(6), 2165-2181, 2013.
- 815 Chang, J., Viovy, N., Vuichard, N., Ciais, P., Campioli, M., Klumpp, K., Martin, R.,
816 Leip, A., and Soussana, J. F. Modeled changes in potential grassland productivity
817 and in ruminant livestock density in Europe over 1961-2010. *PLOS ONE*, 10,



- 818 e0127554, 2015.
- 819 COMIFER. Teneur en P, K et Mg des organes végétaux récoltés pour les cultures de
820 plein champ et les principaux fourrages. Comité Français d'Étude et de
821 Développement de la Fertilisation Raisonnée, Paris, 2007. (in French)
- 822 Cordell, D., Drangert, J., White, S. The story of phosphorus: global food security and
823 food for thought. *Global Environ. Chang.*, 19, 292-305, 2009.
- 824 Cordell, D., Jackson, M., White, S. Phosphorus flows through the Australian food
825 system: identifying intervention points as a roadmap to phosphorus security.
826 *Environ. Sci. Policy*, 29, 87-102, 2013.
- 827 Cordell, D., Neset, T. S., Prior, T. The phosphorus mass balance: identifying 'hotspots'
828 in the food system as a roadmap to phosphorus security. *Curr. Opin. Biotech.*, 23,
829 839-845, 2012.
- 830 Dalin, C., Hanasaki, N., Qiu, H., Mauzerall, D. L., and Rodriguez-Iturbe, I. (2014).
831 Water resources transfers through Chinese interprovincial and foreign food trade.
832 *PNAS*, 111(27), 9774-9779, 2014.
- 833 Elser, J., Bennett, E. Phosphorus cycle: a broken biogeochemical cycle. *Nature*, 478,
834 29-31, 2011.
- 835 FAO. Fertilizer Use by Crop (5th Edition), Food and Agriculture Organization of the
836 United Nations, Rome, 2012.
- 837 Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston,
838 M., Muller, N. D., O'Connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E.
839 M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan,
840 J., Siebert, S., Tilman, D., and Balzer, C. (2011). Solutions for a cultivated planet.
841 *Nat.*, 478(7369), 337-342.
- 842 Grote, U., Craswell, E., Vlek, P. Nutrient flows in international trade: ecology and



- 843 policy issues. *Environ. Sci. Policy*, 8, 439-451, 2005.
- 844 Herrero, M., Havlík, P., Valin, H., Notenbaert, A., Rufino, M. C., Thornton, P. K.,
845 Blümmelb, M., Weissc F., Grace, D., and Obersteiner, M. Biomass use, production,
846 feed efficiencies, and greenhouse gas emissions from global livestock systems.
847 *PNAS*, 110(52), 20888-20893, 2013.
- 848 Levington Agriculture. A report for the European Fertiliser Manufacturers' Association.
849 Levington Agriculture Ltd., Ipswich, UK, 1997.
- 850 Li, J. The development and prospective of nutrition in livestock bones. *Meat Ind.*, 322,
851 41-44, 2008. (in Chinese)
- 852 Liu, Y., Villalba, G., Ayres, R. U., and Schroder, H. Global phosphorus flows and
853 environmental impacts from a consumption perspective. *J. Ind. Ecol.*, 12, 229-247,
854 2008.
- 855 Ma, W., Ma, L., Li, J., Wang, F., Sisák, I., and Zhang, F. Phosphorus flows and use
856 efficiencies in production and consumption of wheat, rice, and maize in China.
857 *Chemosphere*, 84(6), 814-821, 2011.
- 858 MacDonald, G. K., Bennett, E. M., Potter, P. A., and Ramankutty, N. Agronomic
859 phosphorus imbalances across the world's croplands. *PNAS*, 108(7), 3086-3091,
860 2011.
- 861 McDowell, R. W., and Condrón, L. M. Estimating phosphorus loss from New Zealand
862 grassland soils. *New Zeal. J. Agr. Res.*, 47: 137-145, 2004.
- 863 Metson, G. S., Bennett, E. M., Elser, J. J. The role of diet in phosphorus demand.
864 *Environ. Res. Lett.*, 7, 044043, 2012.
- 865 MWPS-18. Livestock Waste Facilities Handbook. Midwest Plan Service, University of
866 Missouri, Ames, IA, USA, 1985.
- 867 Obersteiner, M., Peñuelas, J., Ciais, P., Van Der Velde, M., & Janssens, I. A. The



- 868 phosphorus trilemma. *Nat. Geosci.*, 6(11), 897, 2013.
- 869 OECD Secretariat National Soil Surface Nutrient Balances, 1985 to 1995. Explanatory
- 870 Notes. Coefficients to convert livestock numbers into manure nitrogen quantities
- 871 from national sources. Organisation for Economic Cooperation and Development,
- 872 Paris, 1991.
- 873 Ott, C., Rechberger, H. The European phosphorus balance. *Resour., Conserv. Recy.*,
- 874 60, 159-172, 2012.
- 875 Penuelas, J., Poulter, B., Sardans, J., Ciais, P., Van Der Velde, M., Bopp, L., Boucher,
- 876 O., Godderis, Y., Hinsinger, P., Llusia, J., Nardin, E., Vicca, S., Obersteiner, M.,
- 877 and Nardin, E. Human-induced nitrogen–phosphorus imbalances alter natural and
- 878 managed ecosystems across the globe. *Nat. Commun.*, 4, 2934, 2013.
- 879 Potter, P., Ramankutty, N., Bennett, E. M., & Donner, S. D. Characterizing the spatial
- 880 patterns of global fertilizer application and manure production. *Earth Interact.*,
- 881 14(2), 1-22, 2010.
- 882 Quinton, J. N., Govers, G., Van Oost, K., and Bardgett, R. D. The impact of agricultural
- 883 soil erosion on biogeochemical cycling. *Nat. Geosci.*, 3(5), 311-314, 2010.
- 884 Ringeval, B., Nowak, B., Nesme, T., Delmas, M., and Pellerin, S. Contribution of
- 885 anthropogenic phosphorus to agricultural soil fertility and food production. *Global*
- 886 *Biogeochem. Cy.*, 28(7), 743-756, 2014.
- 887 Sattari, S. Z., Bouwman, A. F., Giller, K. E., and van Ittersum, M. K. Residual soil
- 888 phosphorus as the missing piece in the global phosphorus crisis puzzle. *PNAS*,
- 889 109(16), 6348-6353, 2012.
- 890 Schipanski, M. E., Bennett, E. M. The influence of agricultural trade and livestock
- 891 production on the global phosphorus cycle. *Ecosystems*, 15, 256-268, 2012.
- 892 Scholz, R. W., Ulrich, A. E., Eilittä, M., and Roy, A. Sustainable use of phosphorus: a



- 893 finite resource. *Sci. Total Environ.*, 461, 799-803, 2013.
- 894 Senthilkumar, K., Nesme, T., Mollier, A., and Pellerin, S. Regional-scale phosphorus
895 flows and budgets within France: the importance of agricultural production
896 systems. *Nutr. Cycl. Agroecosys.*, 92(2), 145-159, 2012.
- 897 Sheldrick, W., Syers, J. K., and Lingard, J. Contribution of livestock excreta to nutrient
898 balances. *Nutr. Cycl. Agroecosys.*, 66, 119-131, 2003.
- 899 Smil, V. Phosphorus in the environment: natural flows and human interferences. *Annu.*
900 *Rev. Mater. Sci.*, 25, 53-88, 2000.
- 901 Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M.,
902 Biggs, R., Carpenter, S. R., de Vries, W., de Wit, C. A., Folke, C., Gerten, D.,
903 Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., and Sörlin,
904 S. Planetary boundaries: Guiding human development on a changing planet.
905 *Science*, 347(6223), 1259855, 2015.
- 906 Suh, S., Yee, S. Phosphorus use-efficiency of agriculture and food system in the US.
907 *Chemosphere*, 84, 806-813, 2011.
- 908 USDA-NRCS. Crop Nutrient Tool: Nutrient Content of Crops. United States
909 Department of Agriculture, Natural Resource Conservation Service, Washington,
910 2009.
- 911 van Dijk, K. C., Lesschen, J. P., Oenema, O. Phosphorus flows and balances of the
912 European Union Member States. *Sci. Total Environ.*, 542, 1078-1093, 2016.
- 913 Van Vuuren, D. P., Bouwman, A. F., Beusen, A. H. W. Phosphorus demand for the
914 1970-2100 period: a scenario analysis of resource depletion. *Global Environ.*
915 *Chang.*, 20, 428-439, 2010.
- 916 Waller, J. C. Byproducts and unusual feedstuffs. *Feedstuffs*, 9, 18-22, 2010.
- 917 Wang, R., Balkanski, Y., Boucher, O., Ciais, P., Peñuelas, J., and Tao, S. Significant



918 contribution of combustion-related emissions to the atmospheric phosphorus
919 budget. *Nat. Geosci.*, 8(1), 48-54, 2015.

920 Wang, R., Tao, S., Balkanski, Y., Ciais, P., Boucher, O., Liu, J., Piao, S., Shen, H.,
921 Vuoloc, M. R., Valarie, M., Chen, H., Chen, Y., Cozic, A., Huang, Y., Li, B., Li,
922 W., Shen, G., Wang, B., and Zhang, Y. Exposure to ambient black carbon derived
923 from a unique inventory and high-resolution model. *PNAS*, 111(7), 2459-2463,
924 2014.

925 Wu, H., Yuan, Z., Zhang, Y., Gao, L., and Liu, S. Life-cycle phosphorus use efficiency
926 of the farming system in Anhui Province, Central China. *Resour., Conserv. Recy.*,
927 83, 1-14, 2014.

928 Wu, L., Ma, M. The comprehensive utilization of animals' bone in China. *Modern Food*
929 *Sci. Tech.*, 83, 38-46, 2005. (in Chinese).

930 Xu, J., Liu, X., Wang, F., Zhang, F., Ma, W., and Ma, L. Phosphorus balance and
931 environmental effect of animal production in China. *Acta Ecol. Sinica*, 11, 2911-
932 2918, 2005. (in Chinese).

933