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Summary

The current fertilizer phosphorus (P) made from rock phosphate (rock P) reserves mined each year comes from ever decreasing reserves. This causes continued global increases in the price of P fertilizer, leading to increases in food prices, even though the overuse of phosphorus in fertilizers and manures still causes serious problems in both developed and developing countries. The mean net Chinese soil P surplus in 1980 had increased from 4.6 kg P $^{-1}$ ha to 42.1 kg P ha^{-1} by 2012. In contrast, it was estimated that, in 2017, the UK P balance was positive with a surplus of 6.2 kg P ha^{-1} in managed agricultural land. This had decreased by 0.2 kg/ha (-4%) compared to 2016, and by 3.8 kg ha^{-1}, a decline of -38%, compared to 2000

The overuse of P fertilizer, especially in China and some other developing countries, leads to deterioration of water quality and also cause serious eutrophication. This can lead to the rapid development of

large and toxic algal blooms, in both marine and fresh waters, causing losses of biodiversity and damage to the ecological environment by, for example, harmful invasion of plant species. Mitigating the global expansion of harmful Cynobacterial blooms is a major challenge facing researchers and resource managers. In developed countries such as UK and USA, there are still many problems caused by the overuse of P fertilizers. This applies even in Switzerland although here there have been decreases in nitrogen (N) and P fertilizer applications to agricultural land, which has decreased the N and P loads in Swiss rivers and other surface waters.

There is an urgent wordwide need to alleviate the over use of P fertilizer, Possible approaches are 1) using P fertilizer only when necessary, based upon regular soil P analyses, 2) using different forms of P fertilizer such as slow release products 3) developing new approaches to incorporate this information into nutrient control strategies to decrease the P inputs into water system, 4) drastically increasing the cost of P fertilizer, most likely by taxation, to decrease its use.

Global P fertilizer use

About 75 % of the known rock P (rock P) reserves are located in Morocco (Western Sahara), which is the main exporter of rock P. China and the USA also have significant reserves but it is not sold on the global market, which further limits the supply to other countries (Schoumans et al., 2015). Unlike N, which is obtained by industrial fixation of atmospheric N_2, global P reserves cannot be regenerated and rock P (rock P) reserves are therefore in continuous decline. Unless P can be recycled more effectively, local or even global food crises due to P shortages will eventually occur. Estimates of the time of the effective global depletion of rock P reserves vary from 50 to 100 years (Lewis, 2008: Neseteral et al., 2011) to the next 3 to 4 hundred years (Nature, 2010).

External P inputs to agricultural systems mainly come from mined rock P, animal manures and atmospheric deposition (Lun et al., 2017). About 85% of processed P is used for agricultural fertilizer and as a mineral source for animal nutrition (Dawson et al., 2010). Increased P fertilizer use and livestock production have fundamentally altered the global P cycle. The largest P fertilizer application rates occur predominantly in East Asia, Western Europe, the mid-western United States, and southern Brazil (MacDonald et al., 2011). Inputs of P from the *in situ* weathering of rock and soil particles are insignificant compared with the magnitude of other inputs (Liu et al., 2008). The global application of P fertilizer increased at an average annual rate of 3.2 % from 2002 to 2010 despite a decrease in 2008 that reflected reduced fertilizer application at a time when the price of phosphate fertilizers increased (Cordell et al., 2009,2012). Outputs include P emission into the atmosphere from fires and P loss to uncultivated land or bodies of water (Lun et al., 2017). There is a constant fraction (about 12.5 %) of annual P inputs which is leached or eroded from cropland and pasture soils, independently of agricultural land use type (Bouwman et al., 2013). Phosphorus fluxes from crop residues account for 50% that is recycled in the field, then transformed into livestock feed (25%), and burned or used by other human activities (25%) (Liu et al., 2008; Lun et al., 2017).

Commented [QS1]: I do not know but it form the paper : Global and regional phosphorus budgets in agricultural systems and their implications for phosphorus-use efficiency

Modelling simulations showed that decreases in diffuse nutrient losses of P and N in the Regus area of Switzerland during 1985–2001 were 154 t P and 2609 t N. This is partly attributed to a decrease in agricultural land of about 4 % over this period (Prasuhn et al., 2005). Agronomic inputs of P fertilizer (14.2 Tg of P·y^{-1}) and manure (9.6 Tg of P·y^{-1}) exceeded P removal by harvested crops (12.3 Tg of P·y^{-1}) on a global scale over this period. In contrast, in SubSaharan Africa, in 1993, the annual P loss was 2.5 kg P ha-1 which is unsustainable (Stoorvogel etr al (1993) Phosphorus deficits covered almost 30% of the global cropland area, but only 10% of the global cropland area from 1985–2001, with the largest P deficits contributing 65% to the cumulative global P deficit (Fig.2). In 2014, depending on the P status of the soil, the maximum amount of P that could be applied was 37.1–43.6 kg P ha^{-1} to grassland and 24.0–34.9 kg P ha^{-1} to arable land, The soil P balance in arable land across China, from 1980–2012 showed that the total soil P input was 22.5 kg P ha^{-1} in

1980 and, by 2012, it had had increased to 79.1 kg P ha^{-1} by 2012. However, in 2012, the total soil P outputs only increased from 17.9 kg P^{-1} ha in 1980 to 36.9 kg P ha-1. Therefore, the mean net Chinese soil P surplus in 1980 had increased from 4.6 kg P $^{-1}$ ha to 42.1 kg P ha^{-1} by 2012 (Ma et al., 2018).

In contrast, it was estimated that, in 2017, the UK P balance: was positive with a surplus of 6.2 kg P ha^{-1} in managed agricultural land. This had decreased by 0.2 kg/ha (-4%) compared to 2016, and by 3.8 kg ha^{-1}, a decline of -38%, compared to 2000 (DEFRA, 2018).

The global situation is highly variable. Van Leeuwen et al., (2019) showed that the P input was 35–44 kg P ha$^{−1}$ to grassland on five dairy farms and P inputs were 10–18 kg P ha$^{−1}$, except for one farm which applied 34 kg P ha$^{−1}$ to arable land, while P balances at the five dairy farms were negative (between -2 and -11 kg P ha$^{−1}$), except for the same farm which had a P surplus of 8 kg P ha$^{−1}$. In Australia, the P balances on five dairy farms P balances ranged from -7 to + 133 kg P ha$^{−1}$ (Gourley et al., 2012); Buckley et al. (2015) reported a mean European Union P balance of 6.2 kg ha$^{−1}$ (range from about -100 to + 42 kg P ha$^{−1}$) (Fig.5). In the Netherlands, from 2011–2014 the average P surplus across all dairy farms was about 2.2 kg P ha$^{−1}$ (PBL, 2017). The most widespread large deficits were in South America (particularly Argentina and Paraguay), the northern United States, and Eastern Europe, However, there, 10% of the cropland area with the largest surpluses contributed 45% of the cumulative global P surplus (Fig.1). These large surpluses (which had a median value of 26 kg of P·ha^{-1}·y^{-1}) cover most of East Asia, as well as sizeable tracts of Western and Southern Europe, the coastal United States, and southern Brazil, but less than 2% of the cropland in Africa (MacDonald et al., 2011).

From this data it is concluded that the global distribution of phosphorus soil resources is in imbalance, ranging from considerable overuse to lack, especially in Europe and South America, and critical lack in SubSaharan Africa. Soil P reserves do, indeed, range from feast to famine.

P fertilizer use in UK and China

. The famous Broadbalk Continuous Wheat Experiment at Rothamsted Research, UK, started in 1848 and is still ongoing, Johnston and Poulton, (2018) showed that the same yields could be obtained from appropriate rates of inorganic P fertilizer as from the then main P input, cattle manure. Farmers were thus freed from the very big problem of obtaining and handling bulky and scarce animal manures for fertilizer and could use simple inorganic salts instead. This caused the huge increase in the global use of P fertilizer in agriculture with large increases in crop yields, but also with the correspondingly large environmental problems that we are experiencing today.

A preliminary calculation for the UK suggests that fertilizer P inputs are in the order of 90 kt P per year and crop and animal outputs from farms are about 100 kt P per year (Dawson et al., 2011). In China, the consumption of P fertilizer reached 5.3 Mt P in 2006. (The Yearbook of China Statistics 2007), From 1980 to 2006, the increase in P application rate was 5% per year in China (Li et al., 2011) and only 15–20% of the P applied was taken up by plants in the growing season (Zhang et al. 2008), The accumulation of P in soil as a result of P fertilizer applications to all arable land in China, from 1980 to 2007, was more than 242 kg P ha^{-1} as determined from the balance of P inputs and outputs (Li et al., 2011),

In crop production, large quantities of mineral fertilizers have been used to increase crop yields, with the total amount of mineral N and P use reaching 29.5 and 6.1 Tg by 2010, compared with 9.4 and 1.2 Tg in 1980 (NBSC, 2011). New data concerning phosphate fertilizer consumption were collected from the China Statistical Yearbook (2007–2017) and the China Agricultural Statistical Yearbook (2007–2017) (Fig.3 and 4) (Li et al., 2020). It showed that P fertilizer use in China steadily increased from 7.70 million tons in 2006 to a peak of 8.46 million tons in 2014, after which consumption declined. The application rate of phosphate fertilizer at the provincial level varied greatly between different areas ranging from 12 kg ha^{-1} in Jilin Province to 99. kg ha^{-1} in Xin-jiang, and the mean value was 42.70 kg ha^{-1} (Fig. 2) (Li et al., 2020). The maximum P fertilizer application rate set by China's Ministry of Environmental

Protection is 62.5 kg per ha, while the safe international fertilization rate is 56 kg ha^{-1} (Qinpu, 2014), and most areas in China have not yet exceeded these limits.

The soil microbial biomass can be described as a large pool of potentially available P (c.a. 100 kg P ha^{-1} in UK grassland and much less in arable soils) (Brookes et al., 1984). This P is potentially plant-available but might also contribute to P leaching losses during the process of biomass turnover (Kouno et al., 2002). Application of animal manure cause large increases in microbial biomass P and also accelerate increases in soil Olsen P with a rate almost 3 times higher than when only chemical fertilizer with the same amount of P are applied, probably because much P in manure in highly labile and mineralizable organic forms, which, unlike inorganic P, are only weakly fixed by soil colloids. The desorption of P from soil minerals caused by displacement by organic ligands also contributes to the increase.

Eutrophication in surface waters and sea blue green algal blooms

Phosphorus does not leach through soil readily, unlike nitrate (Dawson and Johnston, 2006), Phosphorus transfer from soil will be principally in the crop and livestock products leaving the farm. It is the losses of P which can occur after these products leave the farm which should particularly lend themselves to managed reduction (Fig. 6) (Dawson et al. 2011). Eutrophication of lakes is caused by over enrichment with nutrients, principally P (Schindler,1997). For example, it is a severe environmental problem in the Baltic Sea (HELCOM, 2014), as well as in many lakes and rivers in its catchment (EEA, 2012). For decades, human activities have caused significant increase in the amounts of N and P entering the sea, causing large and deleterious effects in the ecosystem (Svanbäck et al., 2019),

Diffuse inputs from agriculture are challenging to control, and agriculture is currently the single largest source of anthropogenic nutrient inputs to surface and ground waters. In the Baltic Sea (Fig.7), this amounts to about 40% of total waterborne N and 30% of total P inputs (HELCOM, 2018).

There are several reasons why P removal and recovery from surface waters is important (Rittmann et al., 2011). Firstly, efforts to reduce P inputs to waterways have progressed slowly, and are likely to be insufficient to control harmful algal blooms in the near term (Lürling et al., 2016). Secondly, even if P inputs could be rapidly decreased, soils, sediments and soil organic P have already accumulated sufficient P stocks from prior inputs to maintain aqueous P concentrations at high levels for decades to come (Jarvie et al., 2013; Lürling et al., 2016). Thirdly, recovery of P from surface waters can decrease reliance upon commercial P fertilizer, thereby decreasing the public health and environmental impacts of P fertilizer production and improving food security (Jin et al.,

Commented [QS2]: What is a harmful algal bloom? Harmful algal blooms (also known as HABs) result from the rapid growth, or bloom, of algae that can cause harm to animals, people, or the local environment. They can look like foam, scum, or mats on the surface of water and can be different colors. Harmful algal blooms can occur in warm fresh, marine, or brackish waters with abundant nutrients. There are many organisms that produce harmful algal blooms and harmful algal blooms may have many have different names.

2018).

Rapid blooms of Cyanobacteria, which can cause harm to animals, humans, and the local environment, form foams, scums, and mats on the surface of water These harmful blooms can occur in warm fresh, marine, or brackish waters with sufficient P.. Achieving a P concentration of below 10–20 μg L^{-1}, however, is considered reasonably protective for many ecosystems (Jin et al., 2018), while achieving levels of 1–2 μg L^{-1} can be required to restore oligotrophic conditions in sensitive ecosystems, such as the Everglades (Noe et al., 2001).

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Within coastal zones, harmful algal blooms have expanded geographically and become increasingly common (Hallegraeff, 2010; Glibert et al., 2014; Gobler et al., 2017). Historically, decreasing the P inputs was the strategy applied to control them in freshwater systems (Likens, 1972; Smith and Schindler, 2009). Now, increasing evidence suggests that effective nutrient control strategies must consider and address both N and P inputs (Elser et al., 2009; Lewis et al., 2011). Furthermore, controlling only P inputs in freshwater systems can exacerbate downstream eutrophication in N sensitive estuarine and coastal waters (Finlay et al.,2013).

The aim now is to reduce the N and P inputs into freshwater or marine ecosystems in China, UK and elsewhere and try to develop new approaches to incorporate this information into nutrient control strategies and watershed loading targets to suppress algal blooms. Evaluating the effectiveness of existing physical, chemical, and biological control measures on continued changes in hydrology, stratification, and nutrient dynamics caused by climate change is also required (Paerl et al., 2016).

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