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One Earth

Review

# Understanding soil phosphorus cycling for sustainable development: A review

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## SUMMARY

**Soil phosphorus (P) directly impacts major sustainability outcomes, namely crop yields, water quality, and carbon sequestration. Optimally managing P to improve sustainability outcomes requires a mechanistic understanding of P availability and transfer, alongside high-resolution spatial data. However, it is unclear if current measurement techniques, models, and maps meet the demands for science-informed management. Here, we review recent advances in measuring P fluxes, quantifying P availability, and mapping soil P resources and discuss implications for sustainability outcomes. We find that the understanding of soil P availability has significantly improved but that agronomical applications and climate models are still largely based on outdated concepts. Also, we find that spatial data on soil P resources are highly uncertain, limiting the usefulness of current P maps. We highlight steps to improve existing tools and emphasize that these improvements need to go hand in hand with policy and technological development to successfully address P-related sustainable development goals.**

## INTRODUCTION

Phosphorus (P) has been recognized as a key strategic resource for centuries due to its role as a fertilizer for crops.[1–4] In the 1850s, there was already fierce competition over control of global guano (P-rich deposits of seabird excrements, highly prized as fertilizer) trade, leading the United States to pass the "Guano Islands Act" in 1856, giving United States citizens the right to claim newly discovered guano islands and rocks anywhere in the world as appertaining to the United States.[5] With the discovery of rock phosphate reserves and the industrialization of rock phosphate mining in the early 20$^{th}$ century, P use has increased manyfold.[6] On a global level, current P additions in agriculture exceed crop demands by 9–11 TgP year$^{-1}$ and are projected to increase further until 2050.[7,8] Nevertheless, the strategic importance of P remains due to unequal access to P fertilizers, imbalance in the distribution of global P resources, and large uncertainty about the quantity of P reserves. In addition, the widespread application of P fertilizers has massively altered ecosystem functioning and is widely recognized as one of the main challenges in preserving Earth's habitability for human civilization (Figure 1).[9,10] The influx of N and P to water bodies from agricultural sources leads to eutrophication and has already resulted in coastal hypoxia in several areas.[11] Geological evidence indicates that prolonged high P inputs could even trigger ocean anoxia on a global level.[12] Furthermore, added P stays in the soil for centuries, so past P inputs in agriculture (e.g., from ancient Roman or Amazonian civilizations) continue to influence species distribution and ecosystem productivity even centuries after land abandonment.[13,14]

The seemingly contradictory challenges of P limitation and excess affect several sustainable development goals (SDGs, Figure 2). While in some regions, high P fertilizer use leads to increased P losses that threaten "life below water" and "clean water and sanitation," in other regions, especially in least developed countries, limited access to P fertilizer constrains agricultural yield and is a barrier to the SDGs "no hunger" and "no poverty."[15–17] The SDG "responsible consumption and production" relates to the need to recycle P, as well as to carefully consider justice in P management strategies.[18] In addition, P limitation is a key uncertainty in determining how terrestrial ecosystems react to a changing climate and how much C they will sequester from the atmosphere.[19,20] Improved quantification of C-N-P interactions is thus essential for making science-informed land use decisions that mitigate climate change, relating to the SDG "climate action."[21–24] Finally, distribution of P in soil impacts plant and microbial community distributions and productivity, as these communities have adapted to specific levels of P availability,[25,26] relating to the SDG "life on land."

To reach P-related SDGs, there is an urgent need for soil P data among various stakeholders. For instance, individual

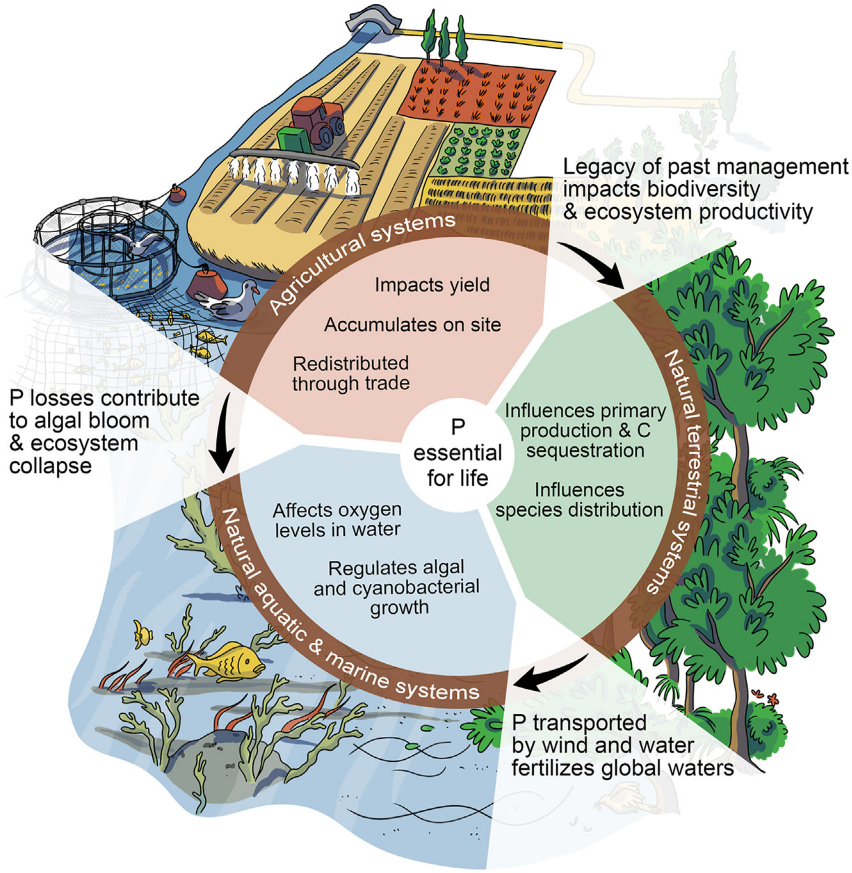


**Figure 1. The relevance of P for ecosystem functioning**

Phosphorus (P) is essential for life, and thus, most ecosystems react sensitively to changes in P availability.

farmers need to know P availability in their soils in order to responsibly and economically apply P fertilizers and manure. Accurate data at appropriate spatial resolution, coupled with robust mechanistic models, are especially relevant to determine P requirements of crops in areas with large yield gaps, such as in sub-Saharan Africa.[15,27,28] At the law and policy level, there seems to be increased attention on balancing crop production with P losses to aquatic and marine systems. For example, in the proposed soil monitoring and resilience directive, the European Commission has recommended monitoring soil-available P as an indicator for nutrient excess and deficiency. This is part of the broader initiative to use P as a key measure of soil health.[29] Implementing and evaluating this indicator requires both accurate spatial data and a sufficient mechanistic understanding of P availability and fluxes. These efforts underscore the need for a combination of detailed geographical information and reliable modeling to effectively manage soil P levels.

The underlying question of this review is how scientific advances in the last decade, namely a massive increase in data availability, harnessed by machine learning algorithms, and the spread of novel measurement and modeling techniques, redefine what we know about P fluxes and the availability of P in soil. Specifically, we ask if the current level of understanding is up to meet demands for addressing P-related SDGs, as well as pinpoint remaining challenges. We begin by synthesizing recent literature to update conceptual models of the P cycle. We then examine recent breakthroughs in understanding the temporal dynamics of P cycling in soil, shedding light on the nuances of P availability. Following this, we critically examine the current state of mapping P in soils. We conclude by identifying key areas where science, law and policy, and technology must converge to effectively contribute to achieving P-related SDGs.

## QUANTIFICATION OF P FLUXES

Estimates of P fluxes at the global scale were already available in the early 2010s and led to the establishment of planetary boundaries for P.[6,9] In the meantime, estimates of P stocks and fluxes have been revised and updated for natural biomes,[30] agriculture,[31] fishery,[32] and global biogeochemical models. However, improvements toward SDGs require policies informed by the situation at the corresponding scales, ranging from plot to national to global scale. Conceptual models of P cycling are the basis for both lab-based research[33–36] and modeling,[37–39] including assessments of implications of P-related policy targets on crop yields.[7,40,41] Despite the relevance of P stocks and fluxes for calibrating models and guiding lab research, a comprehensive review of such data is currently missing. In this section, we bridge this gap by synthesizing the relative sizes of P stocks and fluxes in natural and agricultural terrestrial environments from existing literature, focusing on the plot scale. Though the P cycle is context specific, our analysis of roughly 775 empirical flux measurements from 26 studies supports drawing several general conclusions (Table S1). The general rules as well as the values summarized in Table S1 should be used as guardrails to check the consistency of P model representations.

In both natural and agricultural systems, empirical data on P stocks tend to follow the pattern soil $\gg$ microbial biomass > plant biomass. Soil P stocks tend to be on an order of 100–1,000 g P $m^{-2}$ in the top 50 cm, with only few soils in highly weathered tropical soils having <100 g P $m^{-2}$ and few soils such as on P-rich basaltic parent material or heavily fertilized soils having stocks >1,000 g P $m^{-2}$.[42,43] P in microbial biomass ranges from 8 g P $m^{-2}$ in deserts to 70 g P $m^{-2}$ in boreal forests and tundra.[44] Croplands have, on average, a microbial P stock of 12 g P $m^{-2}$, but large variability exists.[44] Plant biomass tends to be the smallest of the three pools, constituting <10 g P $m^{-2}$ in most ecosystems, ranging from 0.2 g P $m^{-2}$ in semi-desert environments to 29 g P $m^{-2}$ in fertile tropical rainforests.[45,46] Even in agricultural systems, plant biomass tends to contain less P than soil microbial biomass. For example, modern high-yielding corn

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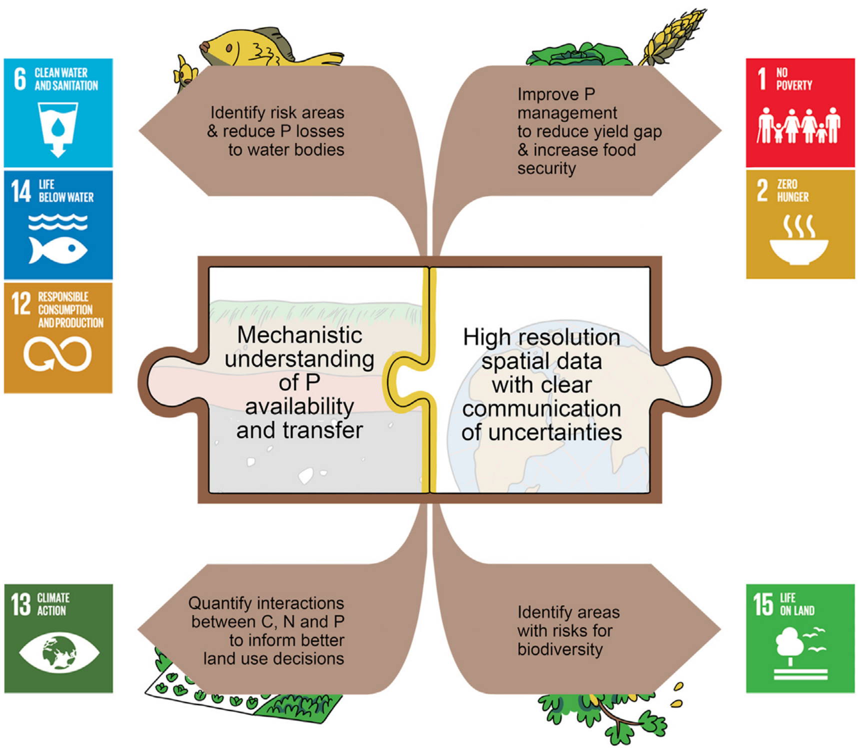


**Figure 2. Science-informed management to reach P-related sustainable development goals**

Improved management of P relates to several sustainable development goals and requires both a better understanding of P availability and transfers and high-resolution spatial data. The numbers correspond to the Sustainable Development Goals of the United Nations.

and soybean contain 2.1 and 4.6 g P $m^{-2}$ at crop maturity, respectively.[47,48] The large size of the microbial P pools combined with faster turnover rates than for plants underline the relative importance of microbes for biological P cycling.[45,49]

In terms of P fluxes, several general conclusions can be drawn. Firstly, recent evidence supports the paradigm that the natural P cycle is fairly closed (Figure 3A).[50] The order of magnitude of P fluxes in the soil-plant system in natural ecosystems decreases from fluxes between P pools within soil $\gg$ fluxes between soils-plants > system inputs (weathering, atmospheric deposition) and losses (erosion, leaching). The P cycle is thus dominated by fluxes within the soil, such as adsorption-desorption and precipitation dissolution (abiotic) and immobilization and mineralization (biotic). These processes have P fluxes on the order of 10 to $10^5$ g P $m^{-2}$ $year^{-1}$ (Table S1). Fluxes within the soil, when extrapolated to a yearly basis, are thus much larger than soil P stocks, implying that some phosphate ions cycle hundreds, thousands, or millions of times between different soil forms within the course of a year.[51] Annual fluxes from soil to plants, called plant uptake, tend to be in the range 0.08–1.8 g P $m^{-2}$ $year^{-1}$.[37,41] A similar range of values has been measured for P returned to soils via litterfall in various ecosystems.[52–54] Natural P inputs and outputs to the soil-plant system tend to be smaller still. P inputs both from rock weathering and through atmospheric deposition range from <0.001 to 0.05 g P $m^{-2}$ $year^{-1}$ in most contexts, with higher values up to 0.1 g P $m^{-2}$ $year^{-1}$ only in montane environments with high weathering rates or in areas downwind of deserts experiencing extreme atmospheric deposition rates.[55–58] Natural losses through leaching and erosion are on similar orders of magnitude as P inputs.[55,58,59] Younger soils tend to have higher P inputs and losses relative to older soils[55,58,60]; however, even in young soils, the amount of P added and lost is small when compared to how much is recycled within the soil itself.

Secondly, human land use, particularly agriculture or high-input forestry, transforms the P cycle from a relatively closed system into one that is substantially more open. Fertilizer application, typically ranging from 1 to 5 g P $m^{-2}$ $year^{-1}$, significantly exceeds natural inputs through weathering or deposition by at least one, and up to four, orders of magnitude (Figure 3C). This leads to fundamental changes in the P cycle, most notably accumulation of actively cycling P in the soil, along with increased losses through erosion and leaching, which are also notably higher than those in natural systems (Table S1).[61–64] Such high-input agricultural P cycles are characteristic of croplands in many regions, including most parts of China, India, South America, and large areas of North America and Europe.[17,65] In regions with limited access to P fertilizers, notably vast areas of sub-Saharan Africa, agricultural practices deplete soil P reserves (Figure 3B). This is because crop harvesting results in substantial P removal (up to several g P per $m^{-2}$ per year[41,47,48,66]), and agricultural activities increase soil erosion, leading to heightened P losses.[61] In many parts of Europe and North America, current P additions are also lower than P removal rates.[65] However, due to the legacy of high P additions in recent decades, soils in these regions still contain significant amounts of anthropogenic labile P.[65,67] In these contexts, high yields can be maintained by tapping into legacy P, i.e., P that has accumulated from high P additions in the past.[68] However, the use of accumulated P in soils may not be straightforward,[69] and these soils may continue to sustain considerable losses through leaching and erosion.

## TOWARD A MECHANISTIC UNDERSTANDING OF P AVAILABILITY FOR AGRICULTURAL APPLICATIONS

While soils contain large P stocks compared to plant biomass, only a tiny fraction (usually <1%) of soil P is in the soil solution, where it can be taken up by plants and microbes. Available P refers to the portion of soil P that can dissolve or be released into the soil solution as phosphate during a plant's growth period and is influenced by the processes of diffusion/desorption, solubilization of P minerals, organic P mineralization, and vertical transport.[70] In general, fertilizer recommendations are based on

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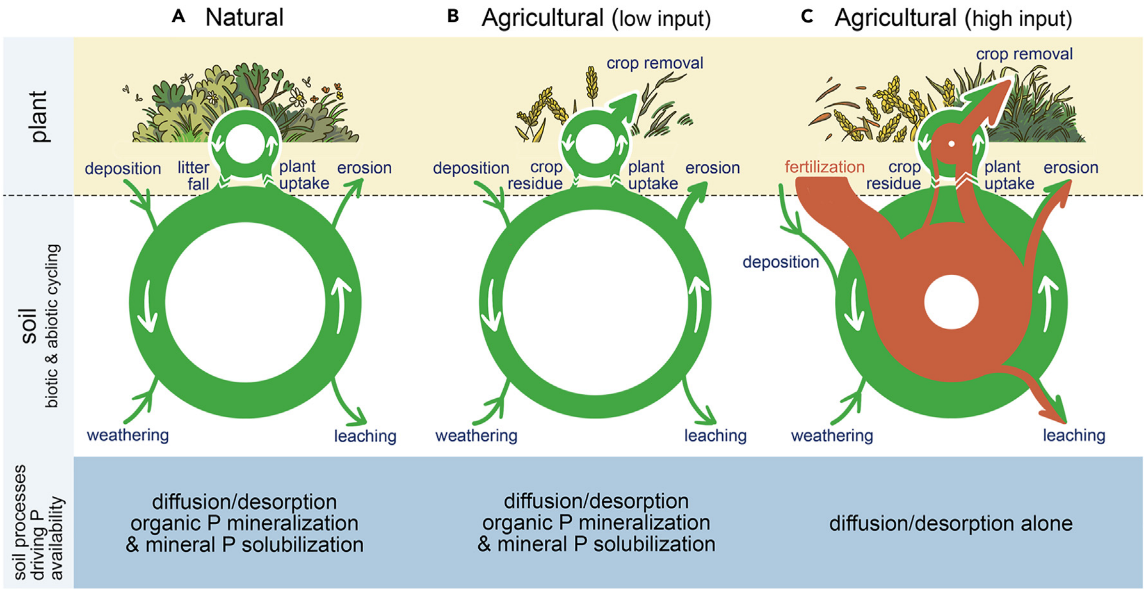


**Figure 3. The terrestrial P cycle in fluxes**

At the plot scale, three conceptual models of the P cycle can be differentiated. In natural terrestrial systems (A), the P cycle is relatively closed with only small inputs and outputs. In no- or low-input agricultural systems (B), P removed by crops and lost through erosion is not sufficiently replaced by inputs, leading to progressive impoverishment of soil P reserves. In high-input agricultural systems (C), P fertilizer inputs lead to a more leaky P cycle with high losses through leaching, erosion, and crop removal and progressive accumulation of P in the soil. The widths of the arrows are based on empirical measurements and modeled values of P fluxes from 27 references, which are summarized in Table S1.

estimating soil P availability using a soil P test and calibrating that result based on regional field trials that measure yield as a function of P test value.[71] The most common soil tests are Olsen P, Bray-1 P, Mehlich-3 P, and Colwell P, but there are many more, and the standard varies from country to country.[72–74] The amount of P extracted by these methods is, first of all, a function of the chemical or mixture of chemicals used. Whereas these extractants solubilize a fraction of P that can be easily measurable by commercial soil laboratories, they do not extract all available P, and they also extract variable proportions of unavailable P, depending on the chemicals used for extraction. This has been shown repeatedly by using isotopic labeling of soils extracted by these chemicals.[75–77]

For the purpose of providing local fertilizer recommendations, the fact that soil P tests are not process based is irrelevant as long as there is a strong relationship between the amount of P extracted and the response of a crop to an application of P fertilizer.[71] However, problems arise when soil tests are interpreted as stand-alone P availability tests, for example, to define global thresholds for P deficiency.[78] This is because crop response to soil test values varies by soil type and environmental context[79–81] (Figure 4A). Also, the correlation between yield and soil test value is sometimes weak, especially for some crops, so fertilizer recommendations based on soil P tests risk over- or under-fertilizing crops.[79] In fact, international cross-comparison studies of P fertilizer recommendations have shown that due to these uncertainties in the methodologies, there are more than 3-fold differences in the P fertilizer recommendations for similar soil-crop situations, depending on the advisory service.[74] Hence, a more mechanistic understanding that goes beyond simplistic P tests is crucial for optimizing P management.

In high P situations, such as most agricultural systems (Figure 3C), diffusion/desorption is the most important process determining P availability and is, in most cases, sufficient to characterize P availability and plant response.[82] Several methods have been developed to measure P flux via diffusion/desorption, including diffusive gradients in thin films and radioisotopic tracing approaches.[35] Both of these methods have shown to be superior at predicting plant response relative to conventional soil tests.[80,83–85] The isotope exchange kinetic approach uses P radioisotope tracing to quantify P availability as a function of time, with the advantage that it is non-invasive, as only carrier-free radioisotopes need to be added to the soil sample.[34,86] Unlike with wet chemistry soil tests, the yield response to P availability as determined by isotope exchange kinetic experiments is consistent across soil types (Figure 4B). Despite the superior predictive power of isotope exchange kinetic and other approaches relative to conventional soil tests, conventional soil tests are still standard in agronomic applications. For example, crop models used to make fertilizer recommendations commonly use Olsen P due to higher data availability.[38,87] Similarly, Olsen P has been used to calibrate precision farming approaches to predict proximal sensing techniques for predicting soil P availability.[88]

Moving forward, P lab research needs to focus on producing more measurements with process-based methods and making those data available to users. For example, of the roughly 41 published studies using the isotope exchange kinetic approach, most measured only a handful of soils.[51] Only Achat et al.[89] systematically studied a broad set of 102 soils. Since such measurements are time, know-how, and resource intensive, another way forward is to make use of the fact that exchange kinetic parameters can be predicted from more cheaply measured soil

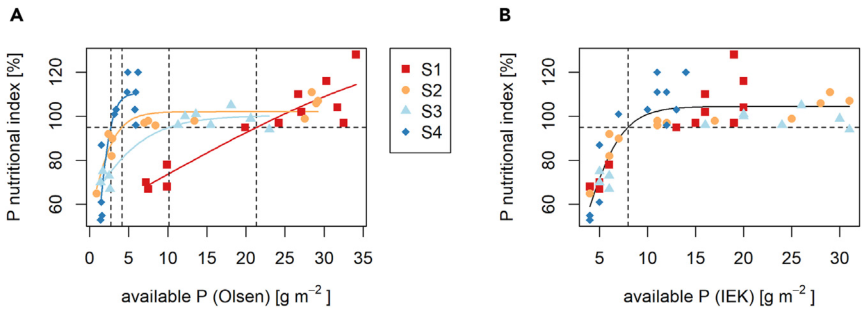


**Figure 4. Plant response to two different methods for measuring available P, illustrated with data from fertilizer trial experiments with corn at four sites in France**
According to Olsen P measurement (A), the available P stock required to reach 95% of plant P nutritional index (dashed lines) varies from roughly 2.5 to 21 g $m^{-2}$, depending on the location. However, using an isotope exchange kinetic measurement of available P (B), P nutritional index follows the same pattern on all sites. This figure was made using data from Morel et al.[80]

properties, namely amorphous Fe/Al oxide, clay, and soil organic carbon concentrations.[89–91] Information on these soil properties allows developing pedo-transfer functions for predicting isotope exchange kinetic parameters. While clay and soil organic carbon are routinely measured, amorphous Fe/Al oxides (e.g., through oxalate extractions), though also cheap, are less commonly measured. More measurement of Fe/Al oxides is thus pivotal for understanding P availability for both agronomic and environmental objectives. In addition, Fe/Al oxides are also key indicators for predicting soil carbon stabilization.[92]

Similarly, users of available P data should evaluate their models against process-based methods where possible. For example, crop models often rely on simple soil P tests to calibrate crop P nutrition (WOFOST (world food studies) based on QUEFTS (quantitative evaluation of the fertility of tropical soils), which uses Olsen P[38]; DSSAT (decision support system for agrotechnology transfer) relies on Bray-1 P combined with exchangeable K[93]) and/or to evaluate modeled labile P pools.[87,94] Given the uncertainties involved with simple soil P tests, we advise future crop models, where possible, to be calibrated with and evaluated against more mechanistic soil P measurements, such as soil solution P concentration and isotope exchange kinetic data. This is particularly important as crop models are used more and more at larger spatial scales (e.g., global grided crop model intercomparison[95]), encompassing a multitude of pedo-climatic conditions, without calibration to each local site condition. Similarly, future developments in proximal sensing of soil P status for precision agriculture applications should be calibrated against process-based representations of P availability rather than simple soil P tests.[88]

## IMPROVING SOIL P REPRESENTATIONS IN GLOBAL LAND SURFACE MODELS

Global land surface models have become increasingly influential tools to predict ecosystem response to global change, such as a warming climate, higher atmospheric $CO_2$ concentrations, and changing nutrient availabilities.[96] The number of global land surface models that include the P cycle has increased significantly over the last decade since the first global land model of C, N, and P cycles.[97] These models have been useful to better understand the interactions between the P, C, and N cycles.[21,39,98] For example, it has been predicted that in the Amazon rainforest, plants allocate 15.3% of their net primary productivity to P acquisition in more fertile sites and 29% in less fertile sites, with important consequences for C sequestration potential.[99] However, parameterization of different processes in those global land models remains quite subjective and is based on very limited data.

Since global land surface models aim at representing all processes relevant for the biosphere's greenhouse gas, water, and energy balance, which operate on a range of timescales (from hours to centuries), complex model representations are needed that account for different fluxes influencing short-term P availability, as well as relatively small inputs and losses such as atmospheric dust deposition and leaching, which are relevant for long-term ecosystem response (Figure 3). To account for the complexity of different processes, most of these models divide soil P into different functional pools (dissolved P, microbial P, adsorbed P, mineral P, organic P).[22,37,97] Empirical data for pool sizes comes from sequential wet chemical extractions, most notably the Hedley extraction.[100,101] Recent research has improved our understanding of the availability of P in these pools and should be used to inform model P representations. For example, using isotopic tracers has revealed that P in the soil solution has a mean residence time in the time frame of seconds to minutes, P that is loosely adsorbed (such as measured with Olsen P and other common soil tests) has a mean residence time in the range of minutes to days, adsorbed P has a mean residence time in the range of days to months, and mineral P has a mean residence time in the range of years to millennia (Figure 5A). Similarly, evidence suggests that the bulk organic P pool as determined by sequential extractions is much more stable than previously thought. Measuring the exchange of P of bulk organic P with the soil solution in a variety of soils over a time frame of months has shown very little exchange, suggesting very long mean residence times of P in this pool.[102,103–107] This is consistent with an analysis of soil organic P in a French cropland, which predicted a mean residence time of 212 years.[108] While some organic P fractions have a shorter mean residence times, playing an important role in P availability, those dynamics are masked in studies of bulk organic P. Such a long and wide range in the mean residence time of organic P is supported by estimates of organic C and organic N turnover,[109] but the fact that bulk organic P is mostly composed of P that is not actively cycling, e.g., due to complexation in organo-mineral complexes,[110] is hardly recognized. For example, analysis of bulk organic P with nuclear magnetic resonance[111] or oxygen isotopes in phosphate[112] needs to consider the relatively inert nature of this pool to avoid drawing false conclusions about P availability and cycling.

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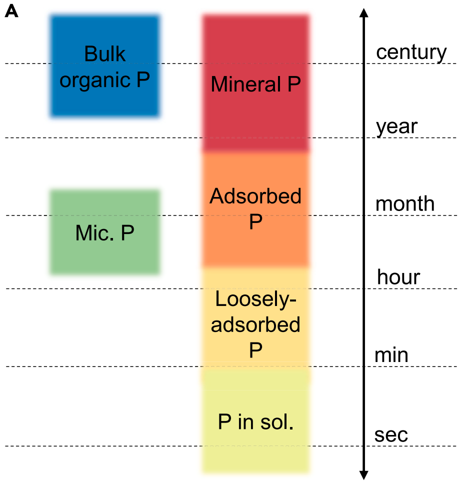


B

| P pool | Corresponding sequential extraction | observed dependencies of MRT | References |
|---|---|---|---|
| Mineral P | HCl-P | longer with increasing pH | (Helfenstein et al. 2020) |
| Bulk organic P | Organic fraction of NaOH-P | longer for soils with reactive soil minerals (assumed similarity with C) | (Raguet et al. 2023; Helfenstein et al. 2021; Doetterl et al. 2018; Torn et al. 1997) |
| Adsorbe d P (NaOH-Pi) | Inorganic fraction of NaOH-P | longer in arable and grassland soils than forest soils; mineralogy also important | (Helfenstein et al. 2020) |
| microbial -P | Hexanol fumigation minus resin-P | longer for low P availability; shortened by drying-rewetting (and other disturbances?) | (Spohn and Widdig 2017; Chen et al. 2021; Oberson and Joner 2005) |
| Loosely-adsorbed P | resin-P, Olsen P | longer in arable than forest and grassland soils | (Helfenstein et al. 2020) |
| P in sol. | $H_20$-P | longer for soils with higher concentration of P in soil solution | (Helfenstein et al. 2018) |

**Figure 5. Estimated MRT of P in operationally defined inorganic and organic soil pools**
(A and B) Mean residence time (MRT) of P pools spans from seconds to > centuries (A). Also, there is considerable variability of MRT between different soils. (B) summarizes reported dependencies of MRT on environmental contexts. Original analysis is based on data from Oberson and Joner,[49] Helfenstein et al.,[51,102,113,114] Raguet et al.,[108] Chen et al.,[115] Doetterl et al.,[116] Spohn and Widdig,[117] and Torn et al.[118]

Second, values of model parameters in the P cycle of global land models, such as turnover rates of adsorbed and occluded P pools, are often globally uniform and/or based on calibration using indirect constraints (e.g., plant productivity gradients instead of P measurements).[22] These problems are, in particular, an issue for soil P processes, such as physicochemical P exchange and phosphatase-mediated mineralization, which control the plant availability of P. Wang et al.[91] deployed a novel inorganic P model based on measurable P fractions, which allowed the assimilation of soil P measurement and demonstrated that values of optimized parameters controlling exchanges differed with respect to soil physicochemical properties and organic C concentration among sites. This finding is in line with evidence from field observations[119] and data synthesis,[120] which indicate that climate and soil properties influence soil P exchange. For example, it has been shown that P in what is commonly termed an adsorbed P pool has a mean residence time in the range of days to months, depending on clay content, organic carbon concentration, and land use.[113] Similarly, the commonly termed primary or apatite P pool, which is often considered to be inert, has been shown to be relatively dynamic in low-pH soils, with a mean residence time similar to the adsorbed P pool.[113] Recently improved global maps of soil P fractions[121] (see Table S2) provide means for the calibration of global models, while global compilations of plant and soil responses to fertilizer addition[2] provide new opportunities for model evaluation. Finally, modeled P cycles should be critically evaluated with empirical evidence of soil P responses to elevated $CO_2$ from decade-long field experiments.[122] Due to the relatively slow turnover of many soil P pools, empirical data on dynamics of different P pools at decadal time scales are particularly relevant for identifying deficiencies in model parametrization. Integrating observation-constrained models of P dynamics into biosphere or agricultural models is expected to increase their reliability and realism, e.g., with respect to P constraints on the biosphere to increasing $CO_2$ or the implications of reduced fertilizer application on crop yields.

## ADVANCES AND PITFALLS OF SPATIAL INFORMATION ON P

In the past decade, spatial data on soil P have increased rapidly. For example, the number of locations with full characterization of P pools (Hedley sequential extraction) increased from 170 in 2013 to 1,857 in 2023 globally.[43,121] A recent compilation of soil P tests (Olsen P and others) identified 33,000 measured samples, a bit more than half of which are from Europe.[72] Together with technological advances, such as machine learning and high-performance computing, and high-resolution environmental covariates from open-source databases and remote sensing products, this has led to a vast increase in the potential of producing P maps. There is high demand for these maps to inform P management and improve food security, reduce P losses, and improve climate and vegetation models (Figure 2), but to what degree are the existing maps up for the task?

We identified 20 soil P maps at the regional to global scale (Table S2). In general, total P has proven easier to predict than other P pools, most notably easier than available P (Olsen P and others). For example, Hengl et al.[123,124] mapped total P and available P in sub-Saharan Africa with $R^2$s of 0.85 and 0.49, respectively. This makes sense given that the total P is highly dependent on environmental drivers such as parent material and climate. However, available P, which is more relevant for sustainability goals, is often more dependent on human management than environmental drivers. Also, the predictability of Olsen P and similar measurements is low because it represents something whose mechanistic definition varies from soil to soil (Figure 4).

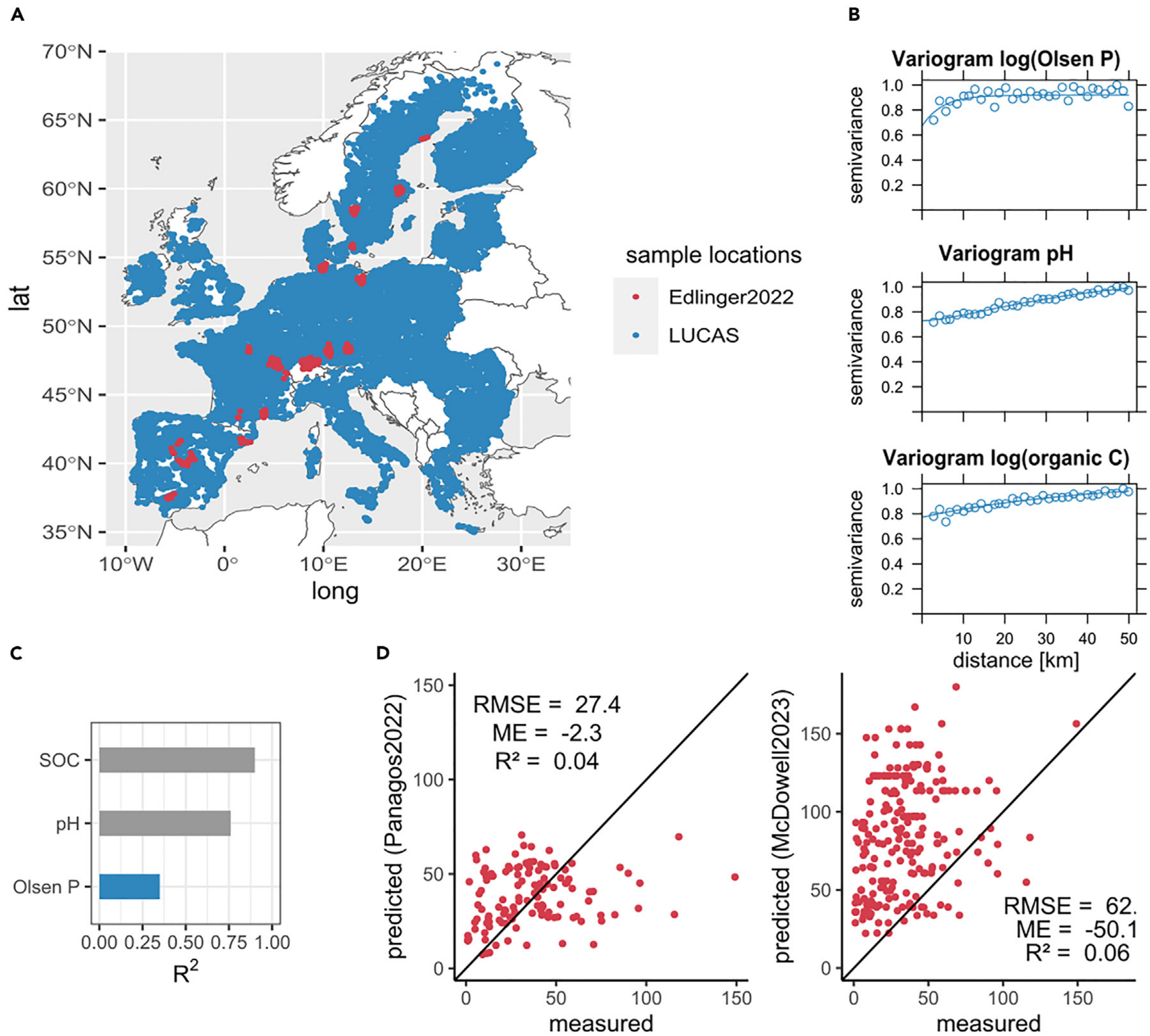


**Figure 6. Challenges for mapping available P, illustrated with the example of Europe**

With $n$ = 17,132, more than half of global available P (Olsen P) data come from the LUCAS dataset (A).[125] The red points show the sample locations of an independent dataset ($n$ = 217), which we used for evaluation.[131] A considerable challenge for mapping is that, compared to other soil properties such as pH and organic C, Olsen P has a low spatial autocorrelation after a distance of roughly 10 km (B). In addition, models (here, random forest) do relatively poorly predicting Olsen P using environmental covariates (C). Independent evaluation of two available P maps[72,125] (D). Original analyses use data from McDowell et al.,[72] Panagos et al.,[125] Edlinger et al.,[131] Fick and Hijmans,[153] Hollister et al.,[154] and Orgiazzi et al.[155] See the supplemental information for details.

We will use the example of Europe to illustrate the opportunities and limitations of current P maps. We focus on Europe because it has the highest density of available P measurements globally (Figure 6A), which have been used to both make European maps[125] and feed into global maps.[72] In addition, Europe is a hotspot of global P pollution due to high (historical) P inputs,[126] and the European Commission aims to use available P maps as an indicator of soil health, proposing that values above 50 mg $kg^{-1}$ Olsen P constitute unhealthy soils.[127] As discussed in the previous section, 50 mg P $kg^{-1}$ Olsen P has completely different meanings in different soils (Figure 4). However, setting that consideration aside, we next evaluate existing maps for informing policymakers on both what proportion of soils in Europe exceeds this threshold and where those soils are located.

The underlying assumptions behind digital soil mapping approaches are that one can predict the property of interest (here, available P) based on the degree of autocorrelation (closer samples are more related than farther away samples) and significant statistical relationships between the property of interest and environmental covariates.[128] Exploring these two assumptions for Europe reveals the inherent challenges of mapping P. First of all, the correlation of individual measurements over distance (autocorrelation) plateaus within several km, much faster than for other soil variables such as pH or soil organic carbon (Figure 6B). This implies that considerable higher sampling density (currently it is around 1 measurement per 220 $km^2$) and statistical sampling techniques adapted toward the intended use of P mapping would be needed to capture the high local variability of available P.[129,130] In addition, available P has a poor correlation with environmental covariates and is thus harder to predict than most other soil properties. The $R^2$ for predicting Olsen P in the European dataset with a random forest model using relevant environmental and anthropogenic covariates (see the supplemental information for details) was 0.35, much lower than for pH or soil organic carbon (Figure 6C). This is supported by soil mapping in sub-Saharan Africa, where available P was the second most difficult to predict of the 19 soil properties analyzed.[123] Evaluating the European available P map[125] against an independent dataset[131] suggests that the bias is small (mean error = $-2.3$ mg kg$^{-1}$), and thus, the map likely gives an accurate approximation of the share of European soils that are above a given threshold (Figure 6D). However, the global available P map[72] has a significant bias, overestimating available P by, on average, 50 mg P kg$^{-1}$, making it not useful for predicting the proportion of soils above a given threshold. An overestimation bias of the global map was also observed in other world regions (Figure S1). Concerning the second objective, knowing where thresholds are overstepped, neither map is useful due to a lack of precision. The high root-mean-square error and the low $R^2$ of both maps show that the map cannot precisely predict where low and high values are found. The poor performance of model predictions is often the case when broad-scale maps are evaluated against independent data.[132] However, errors in predicting available P are particularly high due to the high local scale variability and poor predictability of available P (Figures 6B and 6C) and, ultimately, can be explained by the fact that anthropogenic P input is the key driver of available P[65] and varies from field to field.

Several points would help improve the usefulness of P maps. Firstly, most studies have fallen short of transparently communicating the uncertainty entailed in their maps (Table S2). As a minimum, all P mapping studies should report mean error as a measure of accuracy/bias, root-mean-square error as a measure of precision, and uncertainty maps to show spatial pattern of error.[133] However, of the 20 P mapping studies reviewed (Table S2), only 1 met these criteria.[134] For process-based models, error evaluation is less straightforward because the definition of P pools in the model might differ from commonly available measurement data. Such models should be evaluated against process-based measurements rather than chemical extraction data.[98] Moving forward, maps that show regional probabilities of a soil P threshold being surpassed, rather than point predictions of mean soil P concentrations, would be a more transparent way of dealing with the inherent uncertainties. From a scientific perspective, providing transparency on map uncertainty is important, as it provides indicators where maps need to be improved, either with new sample campaigns or improved mapping methodologies. Providing transparency on map uncertainty is even more crucial for decision-makers, including policymakers and farmers. If map products do not meet quality standards that are needed to fulfill a required purpose, then the consequences can be severe, especially if monetary payment schemes would be coupled to the maps or fertilizer recommendations lead to either reduced yields (too little P) or pollution of surface waters (too much P).

Second, maps need to be more goal oriented, with separate maps for different sustainability objectives. Rather than using one covariate stack to predict all properties of interest, meaningful covariates should be selected for each property of interest depending on scale and context and relying on expert knowledge of P dynamics.[130] As a result, for available P, generic maps for agricultural and non-agricultural soils seem of little use, given the different processes at play (Figure 3). This has already been recognized by some authors publishing separate maps for natural[42,43,121] and agricultural soils.[126,135,136] For mapping P in natural systems, digital soil mapping, relying on environmental covariates, seems appropriate[42,121,137] and can be further improved through additional measurement campaigns and adopting recent technical advances in digital soil mapping.[130] For sustainability goals related to improving P management in agriculture (Figure 2), P maps based on process-based models, which predict soil P pools by modeling P inputs and outputs over space and time, are likely more informative. Process-based P models have recently been developed for agricultural soils in Europe,[40] China,[41] and globally.[126,138] However, here also, efforts are severely hampered by the lack of spatially explicit P input and output data, among others. Such data are usually at the political levels of countries or provinces,[66,125,139] which might detect broad differences between regions but cannot capture field-to-field differences or inform local management per se. More efforts should be made to disaggregate anthropogenic P input and output data, given that data on crop distributions and P management vary significantly among crops.[40,140]

## PRIORITIES FOR TRANSLATING SCIENCE INTO PRACTICE

The previous sections have summarized the current state of scientific knowledge on P in terrestrial systems from different disciplines, as well as suggested future research directions. In this final section, we discuss how scientific advancement is interdependent on technology and policy development to improve P management at different scales (Figure 7).

At the local scale, reducing P input while maintaining (and in some places increasing) agricultural yields requires improved understanding of P availability so that limited P resources can be applied more efficiently.[15,141] This requires close collaboration between industry and science to develop cheaper and more accurate measurements of P pools and fluxes than the currently used soil P tests. These developments lay the knowledge foundation for improved P management, which needs to be followed up with technological development and effective policy incentives. For example, improved P fertilization technologies, such as fertilizer formulation and placement, slow release fertilizers, and variable rate fertilizer application, are needed. There is also considerable potential for reducing P inputs by breeding for crops with higher P use efficiency,[142] which could also benefit micronutrient availability for combating malnutrition.[143] Policy and law should incentivize and/or prescribe adoption of good agricultural practice following research recommendations.[17,88,144] Furthermore, science should work together with industry and policy to develop P footprints for food and consumer products to provide market incentives for higher P use efficiency.

In addition to reducing P inputs, the remaining P has to be recycled better. Whereas agriculture is the largest source of P loss to water in rural areas, in areas with high population density,

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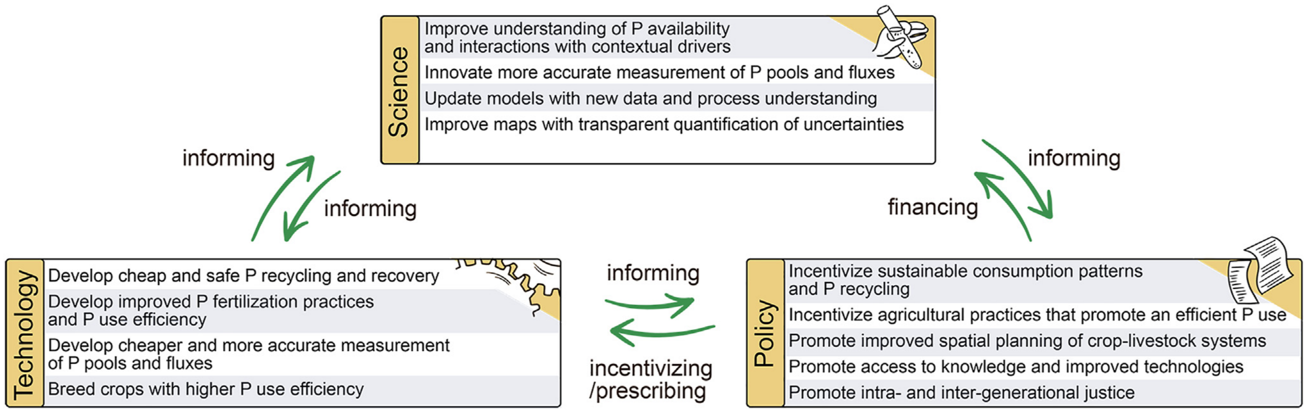


**Figure 7. Solutions and ways forward for science, policy, and technology**

urban waste is the largest P source.[139] Here, increasing efficiency and reducing economic and environmental costs of P recovery from wastewater is a priority. At the policy level, a paradigm shift from P removal to P recovery and recycling is needed.[145,146] At the same time, science and industry need to collaborate to develop cheaper and cleaner chemical and biological processes to regain P from waste.[145,147] However, improved technology alone, even if fully adopted, is unlikely to be enough.[148]

Ultimately, solving the P problem requires cooperative and equitable action at the global scale. Attempts to anchor P management in international governance regimes remain at the level of agenda setting and knowledge exchange, for instance, at the biennial global Sustainable Phosphorus Summit and conferences of the European Sustainable Phosphorous Platform. In the absence of global governance regimes, the management of P supplies essentially falls to unbalanced power dynamics of market mechanisms. This not only threatens the attainment of SDGs but also has severe injustice implications. International law and policy efforts need to improve justice in terms of access to P resources between states and individuals as a means to promote intragenerational justice.[8] This would also severely increase global P use efficiency since low fertilizer application rates to P-deficient soils result in stronger absolute yield benefits than high rates applied to soils with already higher P status.[144] In terms of intergenerational justice, the more P we waste today, the more we endanger future food security, both by polluting fisheries and increasing the scarcity of P fertilizer. However, the share of total imported food and feed P that is recycled has decreased from 30% in 1992 to 25% in 2019.[149] This can be explained by the growing disconnection between crop and livestock production and the rise of landless industrial livestock farms.[150] In the future, agricultural policy should promote improved spatial planning of crop-livestock systems at various spatial scales[149] while acknowledging cultural and economic hurdles.[151] However, reaching planetary boundaries for P input also requires significant dietary change to more plant-based foods.[152] Reducing global P within the confines of safe planetary boundaries thus requires transformational change in various domains. This task presents significant challenges due to the entrenched economic and sociocultural dynamics that underpin current market structures and consumption behavior.[151] Nonetheless, a positive aspect emerges: there is substantial convergence in strategies aimed at meeting other planetary boundaries alongside more sustainable P management.

## SUPPLEMENTAL INFORMATION

Supplemental information can be found online at https://doi.org/10.1016/j.oneear.2024.07.020.

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## AUTHOR CONTRIBUTIONS

Conceptualization, J.H., B.R., F.T., V.L.M., D.S.G., E.A., Y.W., and E.F.; investigation, J.H., X.H., and V.L.M.; writing – original draft, J.H., B.R., F.T., V.L.M., D.S.G., X.H., E.A., Y.W., and E.F.; writing – review & editing, J.H., B.R., F.T., V.L.M., D.S.G., X.H., E.A., Y.W., A.M., and E.F.

## DECLARATION OF INTERESTS

The authors declare no competing interests.

## DECLARATION OF GENERATIVE AI AND AI-ASSISTED TECHNOLOGIES IN THE WRITING PROCESS

During the preparation of this work, the author(s) used ChatGPT (v.4) for individual sentences to suggest individual words to improve language and readability. After using this tool, the authors reviewed and edited the affected sentences. AI was not used to produce content, and the authors take full responsibility for the content of the publication.

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