04

Opportunities for better phosphorus use in agriculture

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Left: A farmer fertilising arable land with granular phosphorus fertiliser. In 2019, around 18 Mt of phosphorus in fertiliser products were applied to agricultural fields and grasslands globally. Photograph courtesy of Adobe Stock.

Low phosphorus use efficiency (~20%) and high phosphorus losses from agricultural land to waterbodies is a growing global problem and exacerbated by climate change and rainfall extremes. Fertiliser use can be optimised and should consider all nutrients. Widespread soil phosphorus testing is required. In some regions appropriate control limits on phosphorus inputs will be needed, whilst in others an increase in P inputs will be required to improve/maintain agricultural productivity. An integrated approach to improve phosphorus use efficiency, reduce losses and increase recycling throughout the food production and consumption chain is needed. A multi-stakeholder approach will, therefore, be critical.

Suggested citation for this chapter: C. Masso, F. Zhang, T.K. Adhya, M.S.A. Blackwell, K.A. Macintosh, P.J. Johnes, P.M. Haygarth, P.J.A. Withers, G. Feng, H. Li, C. Zhang, J. Wu, J. Shen, M.I. Stutter, L. Cheng, W.J. Brownlie. (2022) Chapter 4. Opportunities for better phosphorus use in agriculture, in: W.J. Brownlie, M.A. Sutton, K.V. Heal, D.S. Reay, B.M. Spears (eds.), Our Phosphorus Future. UK Centre for Ecology & Hydrology, Edinburgh. doi: 10.13140/RG.2.2.31727.71846

Challenge 4.1: Low phosphorus use efficiency and high phosphorus losses are common in agriculture

Low phosphorus use efficiency (~20%) and high phosphorus losses from agricultural land to waterbodies is a growing problem globally and is exacerbated by climate change and rainfall extremes. In some cases, slow/controlled-release fertilisers can improve phosphorus use efficiency but these are not yet widely used. In regions where access to phosphorus fertilisers is not a limiting factor, there is a trend to apply high rates of phosphorus to compensate for soil phosphorus fixation, which can increase potential losses. Improving the utilisation of residual phosphorus in soils is critical for achieving efficient agricultural phosphorus use in these regions.

Challenge 4.2: The complexity of soil-crop phosphorus cycles can confound management efforts

The phosphorus cycles that underpin organic, intensive monoculture and mixed farming systems vary widely and are sometimes poorly understood. This can make crop uptake of phosphorus difficult to predict, resulting in inaccurate estimates of fertiliser requirements that may confound attempts to improve phosphorus use efficiency.

Challenge 4.3: Livestock in intensive farming operations are often fed phosphorus in excess leading to high excretion rates

Demand for animal products is increasing. In some regions, poor management (i.e. collection, storage, and application) of animal manures leads to avoidable phosphorus losses to waterbodies. Furthermore, livestock and poultry are commonly fed more phosphorus than they can utilise, leading to excretion of phosphorus-rich manures; they typically retain less than 30% of the phosphorus ingested.

Challenge 4.4: Recycled phosphorus is not sufficiently used in agriculture

A circular approach to phosphorus management in agriculture is critical to address the significant amounts of phosphorus currently lost to the environment or landfills. Recycling is currently limited by transport costs of recycled resources and decoupling of phosphorus cycles across agricultural sectors due to intensification of livestock production. Policies and negative public perceptions about the safety of use can limit phosphorus recycling of certain wastes and residues. Phosphorus recovery technologies can produce contaminant-free phosphorus materials for safe reuse in recycled fertilisers.

Challenge 4.5: There are insufficient policies and targets to deliver integrated action on phosphorus

Policies and/or regulations relating to sustainable phosphorus management at national or regional scales are sparse, and none exist at the global scale. Where regulations exist, policy incoherence and weak enforcement due to the lack of coordination among relevant ministries is commonly observed. Aspirational goals/targets (e.g. for phosphorus recycling, phosphorus losses, phosphorus use efficiency) and indicators to monitor improvement are also lacking for most regions.

Solution 4.1: Provide farmers with the support needed to increase phosphorus use efficiency

Farmers should not apply more phosphorus than needed to maximise crop yields. Fertiliser use can be optimised and should consider all nutrients. Soil phosphorus testing and appropriate control limits on phosphorus inputs may be needed. In some regions, such as parts of Africa, more phosphorus should be applied to improve/maintain crop productivity. Slow-release fertilisers, structural farming measures to reduce erosion and runoff and, innovations to improve uptake of residual phosphorus stores may reduce phosphorus losses whilst maintaining yield. Training farmers and advisors in nutrient management and providing access to decision support systems/tools for nutrient budgeting are required.

Solution 4.2: Implement crop management measures that improve plant uptake of phosphorus in soils

Multiple strategies can be used to optimise phosphorus use efficiency of crops, through site-specific modifications to crop management, integrated soil fertility management (including water and weed management), rhizosphere management and the use of phosphorus efficient cultivars and bio-fertilisers. Strategies can now be developed to improve plant uptake of applied and residual phosphorus in the soil.

Solution 4.3: Optimise animal diets to lower phosphorus excretion and improve manure management

Optimising the diets of animals in intensive farming operations to match growth requirements, and supplementing monogastric animals with phytase enzymes can reduce phosphorus excretion. Governments should provide guidance on recommended dietary phosphorus allowance for livestock based on current scientific knowledge.

Solution 4.4: Increase phosphorus recycling from manures and residue streams

Globally, recycling of treated animal manures and residues and the use of recycled fertilisers should be increased, with corresponding reductions in mineral fertiliser use. Integrating arable and livestock systems can help to reduce costs associated with transporting phosphorus rich animal manures and residues to crops. In some cases, education, extension services and investment in infrastructure and technology are needed to support stakeholders and make phosphorus recycling more efficient.

Solution 4.5: Develop integrated policies and phosphorus use efficiency targets across scales

An integrated approach is essential to increase sustainable phosphorus use in the agricultural sector and will require actions across scales, sectors, disciplines, and regions. Targets to increase phosphorus use efficiency in agriculture and indicators to monitor improvement from farm to global scales are needed. Phosphorus budgets at the farm level are needed to develop catchment management plans that scale phosphorus use efficiency assessments to national, regional, and global scales. We must maximise synergies with other nutrients and ensure that policies are adaptive.

4.1 Introduction

Sustainable agriculture must balance the priorities of environmental health, economic profitability, and social equity, and rests on the principle that our current needs (e.g. short-term economic gain) should not compromise the ability of future generations to meet their own needs (Brodt et al., 2011). Sustainable phosphorus (P) management is an essential component in delivering these priorities. Global agriculture, and subsequently, food security (see Chapter 3), are highly dependent on inputs of P from finite phosphate rock (PR) reserves (see Chapter 2). Most mined PR is used to produce food, with around 85% used for fertilisers, 10% for animal feed, 2–3% for food additives, and the remainder is processed into elemental P for use in a wide range of chemical compounds (de Boer et al., 2019) (see Chapter 2). Accessibility to P resources for agricultural production varies widely between regions, nations, and farms (Cordell and White, 2014). Therefore, whilst many farmers have sufficient access to P, there are instances where 'too little P' or 'excess P' is used (MacDonald et al., 2011).

In a global estimate of agronomic inputs of P in 2000, annual application of fertiliser (~14 Mt P) and manure (~10 Mt P) to soils collectively exceeded P removal in harvested crops (~12 Mt P) (MacDonald et al., 2011). In most cases, P surpluses were the result of the excess application of fertiliser and/or manure. Despite this, almost 30% of the global cropland area was in P deficit (MacDonald et al., 2011) (Figure 4.1).

The elevated risk of P losses from soils receiving P in excess to crop removal is widely acknowledged (Withers et al., 2014a; Huang et al., 2017). Estimated P losses from agriculture to waterbodies vary depending on the modelling approach used.

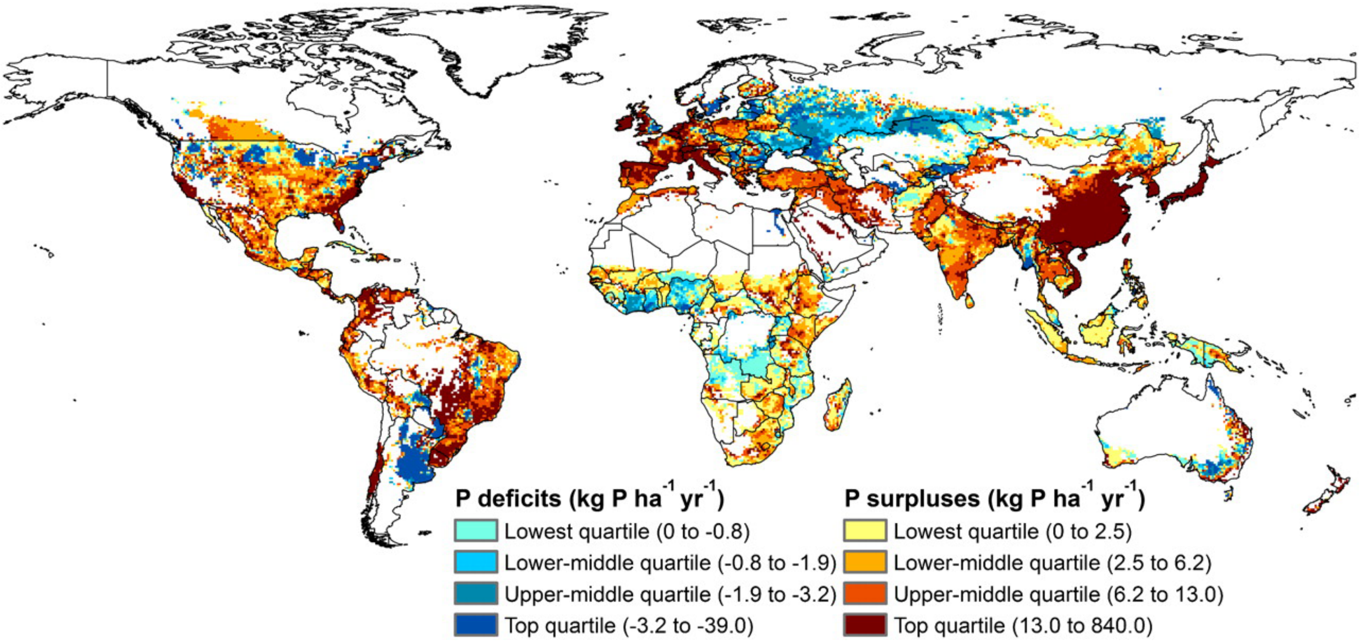

Figure 4.1 Global map of agronomic phosphorus (P) imbalances for the year 2000 expressed per unit of cropland area in each 0.5° grid cell. The P surpluses and deficits are each classified according to quartiles globally (0–25th, 25–50th, 50–75th, and 75–100th percentiles). Figure copyright of MacDonald et al., (2011).

For example, Lwin et al. (2017) estimated that P losses from agriculture in 2010 were 5.7–6.1 Mt P; the estimate of ~11 Mt P for 2013 by Chen and Graedel (2016) was almost double this value. Whilst Beusen et al. (2016) estimate agricultural P losses to surface waters in 2000 were 5 Mt P year^{-1}. These losses are driving the deterioration of aquatic ecosystem health globally (see Chapter 5). In the situation of less P, improving access to enough P fertilisers to increase crop yields and reduce soil P mining is the priority (van der Velde et al., 2014; Filippelli, 2018) (see Chapter 5). Whilst P deficiencies are commonly due to a lack of sufficient P inputs in regions such as East Africa and Brazil, high P fixing soils amongst other soil properties, and a lack of adequate irrigation are also potential constraints on crop productivity (Sanchez et al., 2003; MacDonald et al., 2011). In some cases, options to improve the use efficiency of 'residual P' soil stocks will be required.

The residual P in soils is a measure of the difference between P inputs (e.g. from mineral fertiliser, manure, weathering, and deposition) and P outputs (withdrawal of P in harvested products, and P loss by runoff or erosion) (Bouwman et al., 2009). The accumulation of residual P over time is also known in the literature as 'legacy P' (Kleinman et al., 2011), and resides in soils in a spectrum of plant P availabilities, from labile to non-labile forms depending on the extent of P occlusion in soil minerals and organic matter (Gatiboni et al., 2020). For example, Withers et al. (2001) estimate that since the 1930s, UK soils have accumulated ~12 Mt in legacy phosphorus. Similarly, between 1980 and 2007, soils in China accumulated ~31 Mt of legacy phosphorus (Li et al., 2011) and over 80% of the P in French soils (equivalent to 65 Mt) is associated with past P inputs (Ringeval et al., 2014). Legacy P, therefore, represents a substantial secondary P resource that could potentially substitute for primary inputs of mineral P fertilisers in the short-term, with a large cumulative global influence (Sattari et al., 2012; Rowe et al., 2016). That withstanding, legacy P can also elevate the risk of eutrophication due to the increased transfer of dissolved and particulate P into waterbodies and its accumulation in aquatic sediments (Kleinman et al., 2011; Sharpley et al., 2013; Bingham et al., 2020). Accumulated P can be remobilised or recycled, acting as a continuing source to downstream waterbodies for years, decades, or even centuries (McDowell and Sharpley, 2002) (see Chapter 5). However, the contribution of legacy P in soils and sediments to P loadings to waterbodies over and above P losses from fertiliser and manure applications remains uncertain, and will vary considerably between catchments (King et al., 2017; Stackpoole et al., 2019; Cassidy et al., 2019). Management of legacy P has been discussed in the literature (Kleinman et al., 2011; Sharpley et al., 2013; Wironen et al., 2018; Boitt et al., 2018) (see Chapter 5). In the long-term, a better understanding of P transport pathways within the land–freshwater continuum and climate change impacts on P losses to waters is required. Furthermore, local assessment of the bioavailability of residual P and the length of time this store of soil P can satisfy crop requirements in the absence of primary P inputs is needed.

In this chapter, we provide an overview of the P stocks and flows in agriculture, P dynamics within soils and the co-benefits of improving P sustainability in farming. We then summarise the key challenges and solutions to improving P sustainability in agriculture.

4.2 Phosphorus flows in the global agricultural system

A conceptualisation of the key P flows in the global agricultural system is provided in Figure 4.2. For simplicity, aquaculture and forestry, both of which receive mineral and recycled P inputs, are not included.

In 2019, 18 Mt of P in fertiliser products were applied to agricultural fields and grasslands globally (Jasinski, 2021). Other major P inputs to agricultural soils include animal manures (~12 Mt) and to a lesser extent crop residues (~1 Mt), human wastes (i.e. faeces, urine, wastewater and food waste) (~3.0 Mt) and atmospheric deposition (i.e. P carried in dust and fine soils) (~2.0 Mt) (Chen and Graedel, 2016).

Phosphorus inputs to agricultural soils will either remain in soils (i.e. residual P), be taken up by plant roots, or will be transported away from the soils in runoff, erosion, or, to a lesser extent, leaching to surface waters and groundwater aquifers. Phosphorus in soils can be described as existing in four different inorganic pools, which have varying degrees of availability for uptake by plants (as described in Syers et al., 2008) (Figure 4.3).

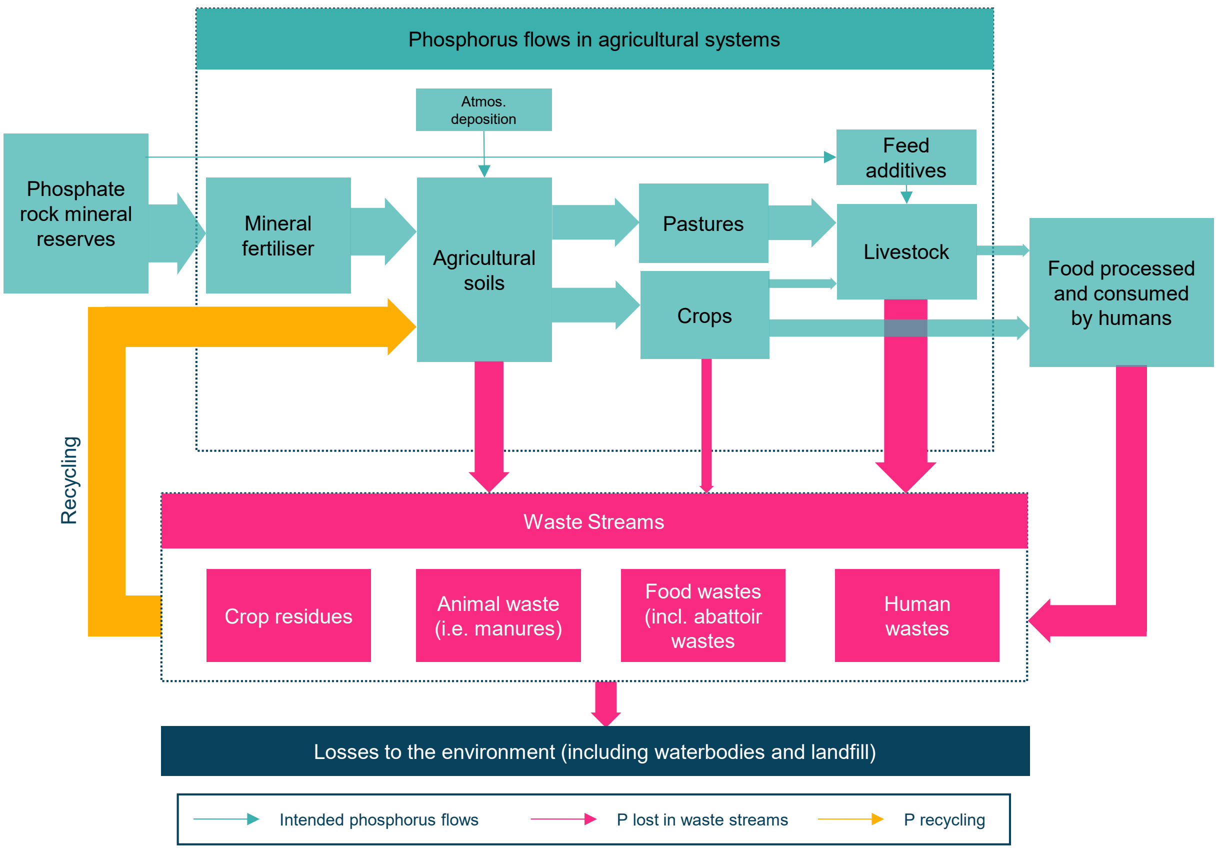

Figure 4.2 A conceptualisation of the phosphorus (P) flows to, within and out of agricultural systems (not including aquaculture and forestry). The width of arrows is proportional to the amount of P estimated in each flow and is based on data from Chen and Graedel (2016). However, wide variation not only exists between nations/farms but also within the flows reported in the literature (particularly for P losses from manure and soils).

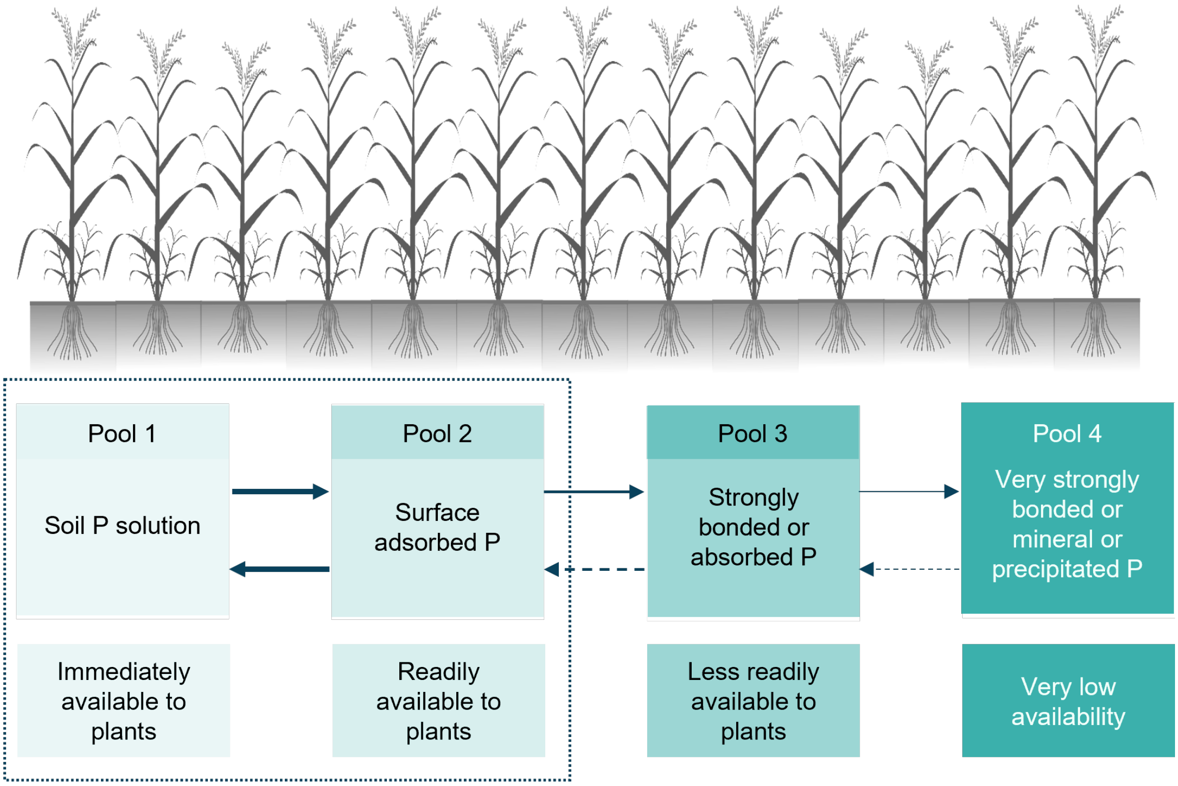

Figure 4.3 Conceptual diagram showing interactions between the forms of inorganic phosphorus (P) in soils categorised in terms of accessibility and plant availability (modified from Syers et al. (2008)).

Pool 1: Describes P in solution that is immediately available to plants.

Pool 2: Describes P bound to the surface of soil particles that is readily available to plants. As the concentration of P in pool 1 is lowered via plant uptake, P in pool 2 is easily transferred to pool 1.

Pool 3: Describes P that is strongly adsorbed to soil particles and is less readily available to plants but can become available under certain conditions and over time.

Pool 4: Describes P that is precipitated or strongly bonded to soil minerals and may only become plant available over many years.

Only 15–25% of the P in fertilisers added to soil remains in solution where it is immediately available to plants, and the remainder is transferred into pools 2, 3 or 4 (Figure 4.3) (Smil, 2000). The P in pool 2 has been shown to provide the bulk of P to plants and, therefore, it is only necessary to accumulate a certain amount of P in this pool to achieve maximum crop yields (Syers et al., 2008). This concept underpins the idea of 'critical P' values for crops, beyond which no increase in yield would be expected (Johnston, 2005). Once the maximum amount of P that can be held in an insoluble form is reached, any additional P applied remains in solution and is available for plant uptake. The P binding capacity of soils is highly influenced by soil type. For example, medium- to fine-textured soils, high in oxides and hydroxides of iron and aluminium, have a high capacity to retain P and are described as P fixing soils. However, in any soil type, P in solution is at an elevated risk of transfer to waterbodies (see Chapter 5).

Therefore, using excessive fertiliser can represent a financial loss to farmers (Sutton et al., 2013).

Losses of P from agricultural soils to waterbodies are transported by lateral surface and subsurface runoff or vertically via leaching to groundwater and/or tile drains, and can result in significant damage to aquatic ecosystems (Chapter 5). It is therefore important that only the most efficient and minimum amount of P is applied to crops (Tirado and Allsopp, 2012). Phosphorus losses in runoff occur when water carries soluble P and particulate P in solution and includes both soluble reactive P and dissolved organic P compounds. The rates of loss are influenced by factors including soil texture, moisture, pH and the P content of the soil, as well as vegetation, and field slope (Mcdowell et al., 2001), while the rate of P release from weathering processes is controlled by pH and P concentrations, both in soils and aquifers (Bingham et al., 2020). The amount of P lost in erosion is influenced by the amount of soil eroded, the soil P content, and soil texture. Diffuse losses from agricultural soils are often low concentration transfers over large areas, from farm fields, and to a lesser extent via mechanical disturbance (i.e. livestock tramping) and wind (i.e. dust storms) (Osmond et al., 2019). Whilst P losses from agricultural soils in tile drainage systems and collecting pipes from drainage ditches (Figure 4.4) are often legislated as diffuse sources (e.g. in the US Clean Waters Act) they can provide focused points of P loading to waterways.

The P assimilated by grasses is either consumed by grazing livestock or cut and fed to livestock, whilst harvested crops are either fed to livestock or humans.

Figure 4.4 Effluent from drainage ditches in a sugar cane plantation in the Everglades Agricultural Area, Florida, USA. Outflow pipes from tile drainage systems and ditches are often legislated as diffuse sources but can provide focused points of phosphorus loading to waterways. Photograph courtesy of Prof. Alan Steinman.

Part of the P in harvested crops is lost in crop residues, some of which is recycled within the agricultural system by ploughing residues back into soils or feeding them to livestock. However, much of the P consumed by livestock is excreted (see Chapter 6). Poor management of livestock manures and slurries can result in high rates of P export from agricultural land to water. Sources of these losses include poorly constructed manure and slurry stores, poor management of wastewaters produced in farmstead operations, fields receiving direct applications of manures and slurries, and animals excreting into rivers and streams directly (James et al., 2007; Lloyd et al., 2019) (Figure 4.5). The P fed to livestock that is not excreted or lost in food processing is passed into products consumed or used by humans (e.g. milk, meat, and fibres). Phosphorus losses in food processing include disposal (often to landfill) of the parts of crops and animals not eaten (~85% of P in mammals is contained in bones and teeth) and food lost through poor storage, distribution, and unwanted goods (see Chapter 8).

Figure 4.5 A cow in Verona, Italy standing in a river with visible algal growth. Direct excretion of livestock wastes into rivers can represent a significant source of phosphorus loading to waters in some catchments, and should be avoided. Photograph taken by Marco Ceschi on www.unsplash.com

Due to losses throughout the food value chain, the amount of P that makes it into the products processed for human consumption (~6.0 Mt year^{-1}) is small in proportion to the P applied to agricultural soils (Chen and Graedel, 2016). An estimated 36 Mt of P was added to agricultural soils in 2013 (~20 Mt P from mineral fertilisers, ~15 Mt P from organic fertilisers (e.g. manures and biosolids) with the remainder from atmospheric deposition and crop residues) (Chen and Graedel, 2016).

4.3 Phosphorus budgets and use efficiency in agriculture

The resources to calculate indicators for P sustainability varies greatly between nations and regions and relies on available data, modelling approaches and expertise. National-scale P budgets are useful to provide a reference for comparison with more detailed indicators, including appropriate chemical and biological monitoring where this can be afforded by countries. Developing national and regional P budgets that sum up the key P inputs and outputs can help to highlight the integration between different components of the P cycle and identify where P losses occur (Chowdhury et al., 2014; Rothwell et al., 2020). Such national nutrient budget activities can also complement the use of local or farm-scale nutrient budgets to help identify excess nutrient use and improve nutrient use decision-making (Öborn et al., 2003; Sutton et al., 2013). Although establishing a direct link between P budget surpluses, losses to water and environmental impact is not straightforward, efforts to reduce P surpluses (i.e. P that does not contribute to productive output) can lessen the burden of P pollution, and improve financial performance in multiple sectors (e.g. in agriculture where P fertilisers and manures are applied in excess of crop needs, or in sectors impacted by P polluted waterbodies).

An often-cited indicator of sustainability derived from P budgets at various scales is ‘phosphorus use efficiency’ (PUE).

Phosphorus use efficiency in animal/livestock production can be considered the conversion ratio of the total P input into useful animal/livestock products (e.g. milk and meat). Similarly, PUE in crop production refers to the conversion ratio of the total P input into useful plant exports (e.g. harvested crops). In cropping systems, this measure of PUE is described as the 'balance' method (outlined in Focus Box 4.1). In an agronomic context, PUE is usually calculated by the 'difference' method which considers not only the P uptake by the crops but also the P removed from the soil (i.e. PUE = (P uptake – P removed from the soil) / P fertiliser applied to soils, where the P removed from the soil is calculated as crop P off-take without any P added). The balance and difference methods can give significantly different measures of PUE (Dhillon et al., 2017), and their benefits in describing P sustainability are discussed in Syers et al. (2008). In addition to using PUE to describe P sustainability in livestock and cropping systems, PUE is also used as a metric to indicate P sustainability in the other components of the P cycle, such as PUE in food processing, and 'full chain PUE'. Full chain PUE describes the P sustainability of the whole food value chain, and can be calculated by dividing net P outputs (e.g. P contained in the food consumed and exported) by the net P inputs (e.g. P in mineral fertiliser, animal feed supplements and food imports). This method has been used to describe full chain nutrient use efficiency and nitrogen (N) use efficiency (Sutton et al., 2013; Rothwell et al., 2020). However, because the definitions for P inputs and outputs and the spatial and temporal criteria can differ markedly between assessments, not all measurements of PUE are comparable. Some considerations when interpreting assessments based on the PUE of cropping systems are provided in Focus Box 4.1.

In recent years, several studies have assessed PUE in crop production (MacDonald et al., 2011; Wu et al., 2016) and crop-pasture production (Hanserud et al., 2015; Özbek et al., 2016) at regional, national and global scales. Studies have also assessed PUE relating to livestock/animal production (Senthilkumar et al., 2012a,b; Chen and Graedel, 2016; Chowdhury et al., 2018) and grassland or pasture-grazing livestock production (Bouwman et al., 2009; Sattari et al., 2016). Livestock systems are the major cause of P inefficiency in regional and national food systems (van Dijk et al., 2016; Withers et al., 2020; Chowdhury and Zhang, 2021) because of the additional P inputs required to produce the large amounts of home-grown feed consumed by animals, particularly ruminants. In a recent global assessment of PUE in agriculture, Chowdhury and Zhang, (2021) showed PUE in the overall agricultural production system (46% averaged across subsystems) was lower compared to the crop-pasture subsystem (averaged as 72%), but higher than the livestock subsystem (averaged as 18%). Whilst agricultural systems differ, poor P management is widespread and a significant cause of avoidable P surpluses and losses (Withers et al., 2020; Chowdhury and Zhang, 2021). Implementing the most effective measures to improve PUE and P sustainability requires an integrated management approach (Cordell and White, 2015a; Sharpley et al., 2018) at the appropriately defined spatial and temporal scale for the system in question.

Focus Box 4.1 - The concept of nutrient use efficiency in cropping systems

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Generally stated, nutrient use efficiency (i.e. commonly referring to P and N use efficiency) is a measure of how much nutrient is taken out of a system relative to the amount supplied to the system. The measurement is quantified based on a defined spatial scale, time period, and system boundary. For example, it can be applied to a field or farm, to a regional watershed, or at a national or global level. The measurement can include all nutrient outputs and inputs, or focus on one part of a system, such as crops or an urban foodshed. When it is applied to a cropping system as a metric of sustainability, it is commonly defined by the mass of plant nutrient in the biomass harvested per unit of nutrient applied and should include all major nutrient sources, regardless of whether they are supplied as mineral fertiliser, manure, or other by-products.

Defined as the equation,

PUE = crop removed/P source inputs

When calculated as a balance of removal to inputs (i.e. using the balance method), it considers only the nutrients removed in the harvested produce, and is therefore referred to as a "partial nutrient balance" (Syers et al., 2008). The balance indicates surpluses or shortfalls but does not provide information on their fate or consequences (e.g. whether surplus P is lost from fields in runoff or is stored in the soils for the next crops).

Since some soils retain most of the P applied, previous management practices influence the soil plant-available P, reflected in a soil test. Agronomic recommendations normally maintain soil test P at or near a critical level at which crop growth is not often limited by P availability. For cropping systems in which soil test P is below the critical level, P input rates greater than crop removal are recommended to increase soil test levels (i.e. a low PUE to raise P soil levels may be desirable in the short term). Where soil test P exceeds the critical level, input rates can fall short of removal rates without reductions of crop yields (Johnston et al., 2014). Thus, the interpretation of PUE depends on other performance metrics, particularly soil test P and crop yield. Low PUE may be desirable in the situations where soils are low in P, but not where soils have sufficient or surplus levels.

PUE is a commonly used metric for nutrient risk assessment, but to provide relevance it must be defined by a system boundary, include a temporal scale, and reference a reliable data source.

4.4 The win-wins of better phosphorus management in agriculture

Measures that reduce P losses and improve PUE in agriculture are a 'win-win', as they aim to increase food production by reducing the need for external P application. This can improve food security, reduce P transfer to waters and associated eutrophication, and, in some regions, reduce costs wasted on the application of excess P fertilisers (including animal wastes). Enhancing PUE and increasing P recycling across sectors will achieve multiple benefits. These include:

Substantially mitigating other pollutant emissions, including reactive N and carbon dioxide (CO_2) emissions to the atmosphere, and N and carbon (C) flux from agricultural production systems to waters. This can be achieved through improved plant productivity and biomass, and consequently sequestration of C and N (Tang et al., 2018). Kirkby et al. (2014) reported a reduction of soil C sequestration under nutrient limiting conditions, including phosphorus. Lorenz and Lal (2010) also reported reduced C sequestration in forest ecosystems under P deficient conditions. However, care is needed as there are situations where optimising production practices to increase PUE may increase the risk of C and N flux from land to water (Zhang et al., 2017a). It will be important to identify those combinations of measures, practices, and influences on farmer behaviour that would deliver multiple benefits (Kanter and Brownlie, 2019).

Boosting the standardisation and development of nutrient-rich waste management for societal acceptance and environmental sustainability of P recycling from waste materials.

Bringing new sustainable economic growth opportunities and development of industrial chains to fertiliser companies associated with innovation of P fertilisers and related novel technologies (see Chapter 7), as well as new business models such as selling 'soil fertility' services instead of fertiliser products (Cordell and White, 2014).

Promoting collaboration between multiple stakeholders involved in different sectors of the whole food system to enhance the full chain PUE.

In the next section, the key challenges in achieving high PUE in agricultural systems are discussed, followed by solutions that will help to deliver a more sustainable use of P in the production of crops and livestock. The importance of integrating the management of soils, crops, and livestock and P recycling into a cohesive P efficient system is highlighted. We conclude with suggestions on how policy and financial support can drive the change needed to build P sustainability into future agricultural systems globally.

4.5 Challenges

Challenge 4.1: Low phosphorus use efficiency and high phosphorus losses are common in agriculture

Low phosphorus use efficiency (~20%) and high phosphorus losses from agricultural land to waterbodies is a growing problem globally and is exacerbated by climate change and rainfall extremes. In some cases, slow/controlled-release fertilisers can improve phosphorus use efficiency but these are not yet widely used. In regions where access to phosphorus fertilisers is not a limiting factor, there is a trend to apply high rates of phosphorus to compensate for soil phosphorus fixation, which can increase potential losses. Improving the utilisation of residual phosphorus in soils is critical for achieving efficient agricultural phosphorus use in these regions.

Low PUE within agricultural systems (i.e. across crop and livestock production) and high P losses from agricultural land to waterbodies are a globally increasing problem (MacDonald et al., 2011; Dhillon et al., 2017; Bouwman et al., 2017). Around 80% of the mined P used in agriculture is stored, wasted, or lost in the food chain between mine, farm and fork (Syers et al., 2008; Cordell and White, 2015b), particularly in areas with surplus P in the soils (Bouwman et al., 2017). The average global PUE calculated between 1961 and 2013 for cereal cropping systems using the 'balance' and 'difference' methods (as described above) produced an estimate of 77% PUE using the balance method, in contrast to 16% PUE using the difference method (Dhillon et al., 2017).

Globally, the application of excess P fertiliser is a greater driver of P surpluses in croplands (>13 kg P ha^{-1} year^{-1}) than manure application (MacDonald et al., 2011), although, in some areas with high livestock densities, manure is an important driver. Furthermore, high P fertiliser application has been typically associated with areas of relatively low PUE (MacDonald et al., 2011).

Currently, in regions where access to P fertilisers is not a limiting factor for farmers, there is a trend to apply high rates of P to compensate for soil P fixation (Ma et al., 2012; Roy et al., 2016; Withers et al., 2018). In recent decades farmers in higher-income countries and China and India have built up significant reserves of residual P in croplands (MacDonald et al., 2011; Bouwman et al., 2017; Zhang et al., 2017b). Residual P can be used by subsequent crops, with many soils now containing sufficient P stores to buffer food security threats for decades (Stutter et al., 2012; Menezes-Blackburn et al., 2018). This is driving a decrease in mineral P inputs in some high-income countries, even leading to negative P budgets in some parts of the EU (van Dijk et al., 2016; Bouwman et al., 2017). Improving the utilisation of residual P in soils is a critical component for efficient P use in agriculture. The challenge is to build agricultural systems that retain and use soil P reserves to grow crops, instead of losing them to waterbodies.

The amount and availability of residual P stored in agricultural soil systems are not always well known (Tian et al., 2017), making it difficult for farmers to know how much P to apply to their soils. The P retention capacity of soils varies globally and impacts the availability of P inputs to crops (Figure 4.6). In P fixing soils, excess P is often applied to overcome P fixation, such as in Brazil (Withers et al., 2018). In this way, residual P in soils plays a dominant role in determining how available P inputs will be to crops (Frossard et al., 2000; Stutter et al., 2012). However, to maximise PUE, fertiliser P inputs must be carefully managed to meet crop demands, whilst taking account of any residual or legacy P stores in soils (Tian et al., 2017). It is important to acknowledge that there are also regions of "too little phosphorus use", such as in parts of Africa, where an increase in P application to soils is required to improve and maintain agricultural productivity (see Chapter 3).

Despite the possibility of exploiting residual P for subsequent crops, surplus P in agricultural soils represents a significant risk of P losses to the environment (Bouwman et al., 2017). In some cases, the use of slow- and controlled-release fertilisers can reduce the risk of P losses (Jones and Oburger, 2011; Teixeira et al., 2016; Fujiwara et al., 2019; Kabiri et al., 2020), whilst bio-fertilisers can improve P uptake of applied and residual P (Adhya et al., 2015; Mukhongo et al., 2017). However, these fertiliser products and technologies are not widely used by farmers. Robust and representative field evidence to support their use in the wide range of agricultural soil types is still required, alongside promotion campaigns to scale up and roll out the application of such novel technologies over large areas.

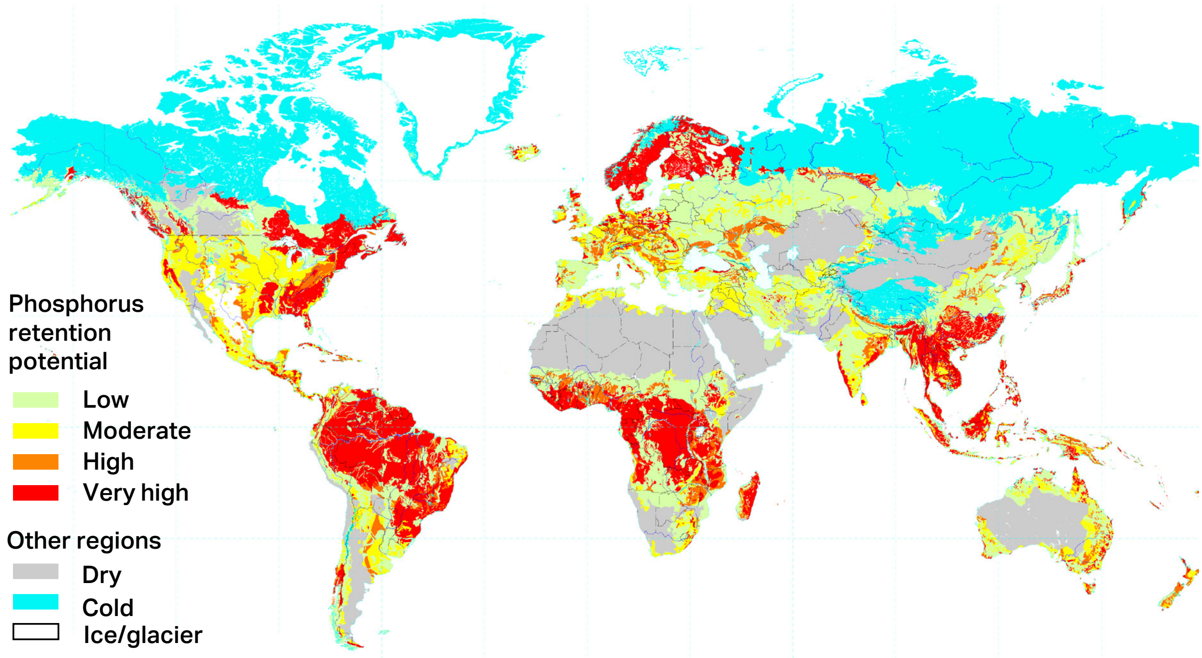

Figure 4.6 Spatial variation in the soil phosphorus retention (adsorption) capacity across the world, indicated by colour, from low (green) to very high (red) (image courtesy of the USDA-NRCS (1998)).

Challenge 4.2: The complexity of soil-crop phosphorus cycles can confound management efforts

The phosphorus cycles that underpin organic, intensive monoculture and mixed farming systems vary widely and are sometimes poorly understood. This can make crop uptake of phosphorus difficult to predict, resulting in inaccurate estimates of fertiliser requirements that may confound attempts to improve phosphorus use efficiency.

The P transfer between soil and plant is influenced by the integrated effects of P transformation, availability, and utilisation caused by soil, rhizosphere, and plant processes (Shen et al., 2011). The complexity of these processes and their interactions (described in Li et al., 2011) makes predicting crop P uptake difficult, which can result in poor estimates of fertiliser requirements (Bünemann, 2015). Indeed, the P cycles that underpin organic, intensive monoculture and mixed farming systems are sometimes poorly understood and can confound P management efforts. Complexity increases with the diversity of organic materials being applied to soils (e.g. manure, sewage sludge, and increasingly new materials with variable composition, such as anaerobic digestate) due to variation in their P content and bioavailability (see Chapter 6). A better understanding of P cycling from organic inputs to soils will help to optimise mineral fertiliser P recommendations for crops and grasses (George et al., 2018).

A key issue to overcome is soil fixation of P, which is the process by which P reacts with other minerals to form insoluble compounds and becomes unavailable to crops (Figure 4.3). The capacity of soils to fix P is highly influenced by the presence of iron, aluminium and calcium, which have peak capacity to fix P at soil pH 3.5, 5.5 and 8.0, respectively (Silva, 2012) (Figure 4.7). It is very difficult to supply sufficient P for crop needs when P solubility is controlled by iron and aluminium. To overcome this, P is commonly applied in excess to crop needs to saturate the soil, however, this can increase the risk of P losses (Withers et al., 2018).

Intercropping (growing of two or more crops together in proximity on the same land) remains widespread in less economically developed countries, especially in South America and Sub-Saharan Africa (SSA), though it has been largely abandoned in more economically developed countries (Bracken, 2019). Where intercropping is practised, one of the challenges is to meet the P requirements of each crop during their respective critical growth stages. However, crops in intercropping systems often have different nutrient and water resource needs at different stages and vary in ability to access the different soil P fractions (Sanyal et al., 2015). In mixed farming systems there is an additional layer of complexity to consider for good P management and cycling. Mixed farming systems imply the integration of crop and livestock farming that must not only manage the P requirements of different crop varieties, cultivars and animal breeds but also optimise the recycling of the different P-rich products they produce (e.g. manure, crop residues, animal residues).

The P demands for each farming activity can be highly variable in both quality (i.e. P bioavailability to different crops, presence of contaminants) and quantity. Understanding how to integrate this information into strategies to enhance PUE in multi-crop systems, particularly for utilisation of residual P stores, is important (George et al., 2018).

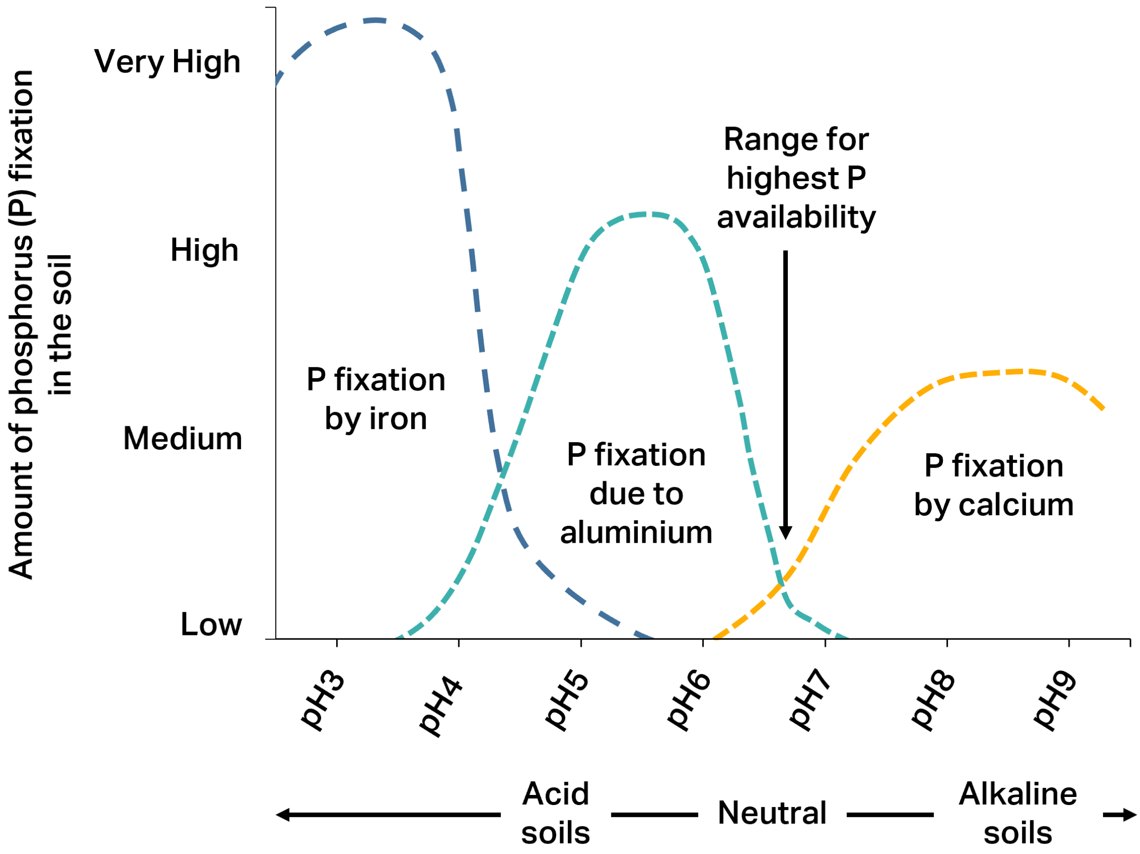

Figure 4.7 General qualitative representation of soil phosphorus (P) fixation (and hence availability) as impacted by soil pH. Modified from Price (2006).

Challenge 4.3: Livestock in intensive farming operations are often fed phosphorus in excess leading to high excretion rates

Demand for animal products is increasing. In some regions, poor management (i.e. collection, storage, and application) of animal manures leads to avoidable phosphorus losses to waterbodies. Furthermore, livestock and poultry are commonly fed more phosphorus than they can utilise, leading to the excretion of phosphorus-rich manures; they typically retain less than 30% of the phosphorus ingested.

Demand for animal products has almost tripled in the last 50 years due to population growth and dietary change (Davis and D'Odoric, 2015). This has placed greater pressure on agricultural systems, driving intensive farming and concentrated production systems (Davis and D'Odorico, 2015; FAO, 2018). Three-quarters of agricultural land worldwide is used for livestock production (Stenfield et al., 2006), and an estimated third of cereal crops are fed to livestock (Alexandratos and Bruinsma, 2012), with this predicted to rise to half by 2050 (Pradhan et al., 2013). Whilst values vary greatly between geographic regions, on average 40% of global crop calories are used as livestock feed; 4.0 kcal of crop products produce about 1.0 kcal of animal product (Pradhan et al., 2013). Livestock numbers are increasing at 2.4% year^{-1} (twice the rate of the human population) to meet this rising demand (Alexandratos and Bruinsma, 2012; UNEP, 2015).

Poor management of animal manures in many catchments, particularly in intensively farmed regions, has led to significant damage to aquatic ecosystems (Stenfield et al., 2006; Oster et al., 2018; Lloyd et al., 2019) (see Chapter 5). This is a key challenge in regions of intensive livestock production, where large quantities of nutrients are imported within animal feed, much of which is then excreted locally in animal manures (Dao and Schwartz, 2011). Structural methods to manage P losses from manures are available at the field and farm scale (see also Chapter 5) and are well documented in the literature (Ulén et al., 2007; Schoumans et al., 2014; Sharpley et al., 2015). In some regions, such as China, direct discharge of animal manures into waterbodies remains widely practised and a significant cause of P pollution (Sattari et al., 2014; Strokal et al., 2016). In many areas of Europe, intensive livestock production has negated some of the improvements in aquatic ecosystems achieved through implementing the European Union's (EU) Water Framework Directive (2000/60/EC) and Urban Wastewater Treatment Directive (91/271/EEC) (Oster et al., 2018).

Monogastric animals and poultry cannot utilise much of the P in their feed because they lack the enzymes to hydrolyse phytic acid, which is an abundant source of P in feed grains (Dao and Schwartz, 2011). Supplementing monogastric diets with phytase enzymes can improve feed digestibility and P uptake (Valk et al., 2000; da Silva et al., 2019). However, livestock and poultry are often fed P in excess of their nutritional requirements, typically retaining <30% of the P ingested (Dao and Schwartz, 2011), leading to excretion of P-rich manures.

The required P supply to animals decreases significantly with increasing live weight and matured skeletal system, and over-feeding with P will lead to unnecessarily high P excretion rates (Poulsen et al., 1999; Oster et al., 2018). A key challenge to nutritional approaches is accurately matching dietary P to the requirements of different species during their different growth stages, without decreasing animal health or diminishing yield of animal products (e.g. meat, milk, eggs) (Lu et al., 2017; Oster et al., 2018). Whilst recommended dietary P allowances are available for livestock and poultry (NRC, 1994, 1998, 2001), Lu et al. (2017) argue that such guidelines are not accurate, particularly for growing and finishing animals, which excrete the greatest amounts of P (Ferket et al., 2002). For example, recommended Chinese guidelines for dietary P to dairy cows are higher than those indicated from studies in Europe and the USA (Guo et al., 2019). Currently, Chinese feeding standards recommend a dietary P content of 0.45% (by weight) for a dairy cow producing 30 kg of milk day^{-1} (MOA, 2004), whilst US guidelines recommend 0.38% for a cow producing 40 kg milk day^{-1} (NRC, 2001). This is despite multiple studies showing that reducing the dietary P content for dairy cows to 0.31% and 0.34% does not affect milk production (Wu et al., 2001; Knowlton and Herbein, 2002; Zhao et al., 2011).

Challenge 4.4: Recycled phosphorus is not sufficiently used in agriculture

A circular approach to phosphorus management in agriculture is critical to address the significant amounts of phosphorus currently lost to the environment or landfills. Recycling is currently limited by transport costs of recycled resources and decoupling of phosphorus cycles across agricultural sectors due to intensification of livestock production. Policies and negative public perceptions about the safety of use can limit phosphorus recycling of certain wastes and residues. Phosphorus recovery technologies can produce contaminant-free phosphorus materials for safe reuse in recycled fertilisers.

A circular approach to P management in agriculture is critical to the delivery of a sustainable P future. Recycling P-rich organic materials and recovering P from waste streams for reuse in new products are discussed in detail in Chapters 6 and 7. A brief overview of some of the challenges of recycling P in agriculture is provided below.

The challenge for recycling P within the agricultural sector is to increase access to secondary P resources and to support the development of policies and regulations that de-risk the use of these resources from farm to global scales (Owen et al., 2010). Phosphorus can be recycled from various wastes, including wastewater, biosolids, municipal wastes, crop residues,and animal by-products, among others (Leinweber et al., 2018). However, P-rich organic materials in waste streams are commonly treated as waste rather than as a source of P input to support production. As a result, these P-rich organic resources are often not collected, stored, processed, or applied effectively, or are applied as a waste to crops and grass to avoid over-full slurry/manure stores, in excess of P requirements, leading to significant P losses to soils, waters or landfill (see Chapter 6).

The intensification of livestock production has enhanced the decoupling of P cycles between sectors. Transporting manures to sites where they can be applied sustainably to land is often not economically feasible, due to distance and the weight and volume of the manures (see Chapter 6). In the case of significant livestock production, the amount of animal manure generated could exceed the P capacity of the receiving soils, particularly when soil fertilisation policies are enforced, i.e. a maximum amount of P that can be applied to a unit area, based on soil testing or plant tissue analysis (Blackwell et al., 2019). The excess manure, when not properly handled, consequently reduces the overall PUE when the full chain is considered from farm to fork (Risse et al., 2006; Lun et al., 2018). Manures produced globally in 2013 contained an estimated 15 to 20 Mt P, of which between 8.0 and 12.0 Mt were recycled back to croplands (Chen and Graedel, 2016; Bouwman et al., 2017).

The variable concentrations and bioavailability of the P contained in P-rich organic materials can also restrict their reliability as a viable fertiliser. Whilst typically lower than mineral P fertilisers, the concentration and bioavailability of P in organic materials are not easy to determine quickly, representing a challenge for farm-scale nutrient management. The bulky nature of many P-rich organic materials can make them difficult to spread consistently, also affecting their perceived reliability as a fertiliser to be used in place of mineral P fertilisers (see Chapter 6). Some manures and P-rich organic materials may also contain contaminants, for example, pathogens, hormones, antibiotics, potentially toxic elements, and micro-plastics, which can accumulate in soils after manure/biosolid application and potentially compromise food quality for human consumption (see Chapter 6).

In some cases, P and other nutrients must be 'recovered' and detoxified from wastes, to recycle them safely and effectively. A further set of challenges, including policy and economic barriers, require addressing to implement P recovery (see Chapter 7). An essential driver of P recovery (and recycling) is the presence of a market for P recovered materials. There are markets for niche recycled fertilisers sold at a small scale (e.g. struvite). However, a potentially significant market option would be to produce contaminant-free P raw materials that can be used by the mineral fertiliser industry as an alternative to phosphate rock. However, this relies on significant industry transformation and support, which may require policy-based motivation (see Chapter 7).

Farmers may choose not to use some P-rich organic materials (e.g human excreta) as fertilisers because of negative perceptions over the safety of their use in food production, and/or policy barriers. Quality standards for specific use, for instance in food or feed crops, could limit the opportunities for use in the agricultural sector. Evaluation based on scientific evidence is therefore required to minimise unnecessary limitations on the use of recycled phosphorus. However, additional limitations for recycling P from some P-rich organic materials could be related to cultural barriers (Mariwah and Drangert, 2011; Andersson, 2015), including the 'yuck factor' (i.e. disgust generated by an aspect of an idea) (Ghernaout et al., 2019; Ricart and Rico, 2019). This is exemplified by a study of a peri-urban farming community in Ghana, which found residents accepted that excreta when appropriately treated can be safely used as a fertiliser, but were not willing to use it on their crops or consume crops fertilised with treated excreta due to perceived health risk concerns (Mariwah and Drangert, 2011). Andersson (2015) argues that such social norms and cultural perceptions should be recognised, but not be treated as absolute barriers to the uptake of P recycling practices.

Challenge 4.5: There are insufficient policies and targets to deliver integrated action on phosphorus

Policies and/or regulations relating to sustainable phosphorus management at national or regional scales are sparse, and none exist at the global scale. Where regulations exist, policy incoherence and weak enforcement due to the lack of coordination among relevant ministries is commonly observed. Aspirational goals/targets (e.g. for phosphorus recycling, phosphorus losses, phosphorus use efficiency) and indicators to monitor improvement are also lacking for most regions.

As highlighted in the challenges above, improving sustainable P management in agriculture will require action across scales, sectors, disciplines, and regions, and cooperation between multiple stakeholders and communities. As acknowledged in the literature (Withers et al., 2014a, 2015; Cordell and White, 2014; Blackwell et al., 2019), an integrated approach is essential to develop and implement strategies that can deliver long-lasting and significant improvements to PUE in the agriculture sector. However, indicators to monitor improvement are lacking in most regions. Where regulations exist, policy incoherence and weak enforcement due to the lack of coordination among relevant ministries is commonly observed. Policies and/or regulations relating to sustainable P management at national or regional scales are sparse, and none exist at the global scale (see Chapter 9). In the EU, relevant policies include the 'Fertilising Products Regulation' (European Parliament, 2019) and the 'EU Critical Raw Materials List' (which has included PR and elemental P since 2017; European Commission, 2017). In Africa, 'The Abuja Declaration on Fertiliser for an African Green Revolution' of 2006 called for the elimination of all taxes and tariffs on fertilisers and outlined targets to increase fertiliser use (African Development Bank, 2021). In other regions, measures that address P sustainability are contained within broader policies and regulations (e.g. 'The Clean Water Act' in the USA; US Government, 1972), or 'The Action Plan for Zero Growth in the Application of Fertilizer' in China referring to chemical fertiliser (MOA, 2015), many of which do not reference P directly, or are based on volunteer schemes and subsidies.

Whilst there are extensive academic publications on sustainable P management in agriculture, government-endorsed guidance and guidelines are lacking in most regions. Although in some regions (e.g. North America, Europe and Australia), selected guidelines for effective use of P inputs to optimise crop and energy production and minimise pollution have been developed and operationalised (Shober and Sims, 2003; Elliott and O'Connor, 2007; Schindler, 2012; Metson and Bennett, 2015). In other regions, such as SSA, such policies and their implimentation are lacking (Masso et al., 2017). Even though these tools exist in areas like North America, Australia and Western Europe, P pollution remains a significant problem (see Chapter 3), suggesting guidelines are ineffective or not properly enforced, or both.

4.6 Solutions

Solution 4.1: Provide farmers with the support needed to increase phosphorus use efficiency

Farmers should not apply more phosphorus than needed to maximise crop yields. Fertiliser use can be optimised and should consider all nutrients. Soil phosphorus testing and appropriate control limits on phosphorus inputs may be needed. In some regions, such as parts of Africa, more phosphorus should be applied to improve/maintain crop productivity. Slow-release fertilisers, structural farming measures to reduce erosion and runoff and, innovations to improve uptake of residual phosphorus stores may reduce phosphorus losses whilst maintaining yield. Training farmers and advisors in nutrient management and providing access to decision support systems/tools for nutrient budgeting are required.

Extensive soil P testing can help farmers manage P applications more effectively (Dhillon et al., 2017). Farmers should not apply more P to soils than needed to optimise crop yields. In some instances, appropriate control limits on the application of P fertilisers may be needed (both from recovered and mineral P sources), especially where bioavailable soil P concentrations are in excess of crop requirements. Shifting from broadcast methods of fertiliser application to more precise mineral fertiliser and manure placement can help maximise plant uptake whilst minimising losses (Withers et al., 2014b; Dhillon et al., 2017). The 4R and 4R plus nutrient stewardship approaches provide a framework to optimise fertiliser and manure use whilst maintaining and improving crop yield, based around the concepts of Right fertiliser source, applied at the Right rate, the Right time and in the Right place (for more details see: Johnston and Bruulsema, 2014; The Fertilizer Institute, 2017). The 4R plus nutrient stewardship approach combines the 4R nutrient stewardship approach with conservation practices or integrated soil fertility management (e.g. reducing tillage, planting cover crops, and adding structures such as contour strips and stream buffer strips among others) (for more details see the Nature Conservancy, 2021). Both approaches require a good understanding of the science underlying nutrient use in farming systems, as well as local conditions in the environment. Training farmers and advisors in nutrient management and providing access to decision support systems and tools for nutrient budgeting are required to support the uptake of such approaches.

In some less economically developed countries, insufficient use of fertilisers and soil erosion has led to substantial nutrient depletion of soils, constraining agricultural productivity, especially impacting marginal and smallholder farmers (see Chapter 3). In regions of insufficient P, opportunities to improve access to P include access to credit, extension services, investment in sustainable infrastructure (such as local P recycling systems from food waste and sanitation), and knowledge exchange to support better PUE and recycling within the agriculture sector (see Chapter 3). Indeed, the recycling of treated animal manures and residues (e.g. bones, blood) as sources of P and the use of recovered P fertilisers should be optimised in all regions, with corresponding reductions in mineral fertiliser use (see Chapters 7 and 8).

In all instances, strategies to improve PUE should consider all nutrient inputs returned to the soils, including those from human waste streams, manures and crop residues (see Chapter 6), and ensure that other crop nutrients (e.g. N, potassium (K) and other micronutrients) are sufficiently available to maximise plant P uptake (MacDonald et al., 2011; Bouwman et al., 2017). Micronutrients are essential for crop growth, and critical components of healthy human and animal diets. Micronutrients are non-renewable, and in some regions scarce, and should be recycled as part of any integrated nutrient sustainability strategy (Bell and Dell, 2008; de Haes et al., 2012; Mensink et al., 2013; Vaneeckhaute et al., 2019). Long-term P management planning at the farm scale needs to involve soil P dynamics to elucidate P budgets, taking into consideration the agronomic value of residual P (Powers et al., 2016; Sharpley et al., 2018). This calls for better diagnostic tools to determine the distribution and plant availability of residual P stores (Blackwell et al., 2019) and the adoption of a cumulative PUE indicator. Strategies to improve plant uptake of residual P could allow a reduction of P inputs to some soils, and reduce the risks of P losses to the environment (Stutter et al., 2012, 2015; Menezes-Blackburn et al., 2018; George et al., 2018).

Reducing diffuse losses of P from agricultural soils is a key component in strategies to improve P sustainability, and can be a win-win, with benefits to both the farmer and the environment. Historically, studies of diffuse P losses have focused on the transport and distribution of P in the surface soil layers, due to the general assumption that vertical transport was relatively insignificant due to the high P fixing capacity of most subsoils (Gburek et al., 2005). However, more recent field studies have shown that P export via subsurface flows to surface waters and groundwaters can also be significant in soils receiving continual fertiliser application in excess of crop requirements which result in P accumulation, especially in those soils that are P saturated, or have low P retention capacity (Szogi et al., 2012; Boitt, 2017; Tian et al., 2017).

Diffuse P losses can also be impacted by irrigation. In a long-term irrigation trial of soils under grazed pasture, a three-fold increase in irrigation frequency resulted in a 13-fold increase in P loss in irrigation outwash (Boitt, 2017). If diffuse P losses are minimised, residual P can represent a long-lasting source of P to subsequent crops (Syers et al., 2008; Johnston and Poulton, 2019). Common practices to reduce diffuse P losses include land and soil management that reduce soil erosion and control the drainage rate or filter drainage (Ulén et al., 2007; Schoumans et al., 2014; Sharpley et al., 2015; see a summary of measures in Chapter 5). Mitigation options need to be informed by the identification of loss and/or inefficiency hotspots or events (Haygarth et al., 2005; Senthilkumar et al., 2012a). The efficacy of erosion control practices, such as a reduction in tillage, are impacted by soil type, climate, landscape and land management practices, and should be appraised for their positive and negative effects on PUE at the catchment and farm levels (Ulén et al., 2010). Vegetated buffer strips between cropland and watercourses are promoted as a principal control measure for diffuse P transport and can reduce runoff velocity, trap sediments, increase infiltration, and, ultimately, increase plant uptake of nutrients (Dorioz et al., 2006; Roberts et al., 2012; Kieta et al., 2018). However, continual management, such as harvesting of vegetation and control of soil redox conditions, may be required to ensure that buffer strips continue to effectively reduce P transfer to rivers (Stutter et al., 2009; Johnes et al., 2020).

Innovations in fertiliser technologies can be utilised to decrease P losses, for example particle surface coating technologies that control the release of P to plants (Everaert et al., 2016; Teixeira et al., 2016; Bernardo et al., 2018; Fujiwara et al., 2019; Ramírez-Rodríguez et al., 2020; Kabiri et al., 2020; Qi et al., 2020). Such technologies can help reduce fertiliser requirements, and P losses to surface waters and groundwaters via erosion, surface and subsurface flow pathways. That withstanding, in many soils, struvite ($MgNH_4PO_4{\cdot}6H_2O$), which can be recovered from wastewaters, can be promoted as an efficient slow-release P fertiliser (Kataki et al., 2016; Schipper, 2019; see Chapter 7).

Solution 4.2: Implement crop management measures that improve plant uptake of phosphorus in soils

Multiple strategies can be used to optimise phosphorus use efficiency of crops, through site-specific modifications to crop management, integrated soil fertility management (including water and weed management), rhizosphere management and the use of phosphorus efficient cultivars and bio-fertilisers. Strategies can now be developed to improve plant uptake of applied and residual phosphorus in the soil.

Farmers can optimise the PUE of crops by optimising plant spacing (Venkatesh et al., 2019) and planting times (Mukherjee et al., 2017), and selecting appropriate intercropping and crop rotations (Bationo and Kumar, 2002; Darch et al., 2018). For example, intercropping maize and faba bean, maize and chickpea, wheat and common bean, and clover and barley has been shown to improve PUE (Li et al., 2003, 2007, 2008; Darch et al., 2018). Water, weed and pest management can improve crop PUE, as part of integrated soil fertility management planning (Bationo and Kumar, 2002; Blackwell et al., 2010; Vanlauwe et al., 2010). For example, novel irrigation systems such as drip irrigation, partial root-zone drying irrigation, and fertigation can enhance water use efficiency and PUE (Yactayo et al., 2013). Control of root borne diseases is essential to ensure a healthy root system for efficient P uptake into crops (Altieri et al., 2012). Weed control minimises competition for soil P resources between the target crop and weeds (Naragade et al., 2018); hence, in principle, weed-free crops will require fewer nutrients than weed-infested crops.

Opportunities exist to optimise P management at the zone of interaction among plant roots, soils, and soil microorganisms (i.e. rhizosphere P management) (Shen et al., 2011; Richardson et al., 2011). Soil pH is one of the key factors governing soil P bioavailability, with soil P typically most bioavailable at pH 6.0-7.0 (Hinsinger, 2001; Silva, 2012) (Figure 4.6). Localised application of ammonium and P fertilisers in calcareous soils can decrease soil pH by up to 3 units (i.e. 1000-fold), and stimulates root proliferation in maize leading to improved PUE and plant growth (Jing et al., 2010). The addition of lime to soils can help reduce aluminium toxicity and subsequent damage to roots and improve the PUE of some cropping systems (Syers et al., 2008). However, considerable contradictions exist in the literature regarding the impact of soil liming on soil pH and soil P availability (Syers et al., 2008). Interpretation of studies examining the impact of pH on plant P uptake should be treated with caution since pH has a profound impact on factors other than soil P solubility (Penn and Camberato, 2019). Farm-scale assessments are, therefore, recommended to inform liming practices with respect to soil P availability.

Amending soils with biochars, composts, manures and poultry litter can drive a reduction in P adsorption to soil particles, and/or change soil pH, which can also alter soil P availability (Shen et al., 2011; Ch'ng et al., 2014). However, the potential to improve plant access to residual P through manure additions (Shen et al., 2011) and organic amendments requires further research and is likely to be site-specific (Penn and Camberato, 2019).

The use of P-efficient plant cultivars with higher P acquisition capacity can lead to more efficient utilisation of soil P pools (Simpson et al., 2011; Heuer et al., 2017). Root architecture plays an important role in maximising P uptake and modifications in root architecture in response to P deficiency are well documented (Niu et al., 2013). Root systems with high surface areas, that extend into P-rich soil zones, can access P in a given volume of soil more effectively (Lynch, 1995). Selecting genotypes with high root foraging capacity (e.g. more adventitious roots, lateral branching, or shallow roots) to enhance P uptake can significantly reduce P fertiliser input requirements and P losses. For example, the P-efficient genotypes of the common bean have more shallow roots in the topsoil where there are relatively more P resources (Lynch and Brown, 2008). Selecting genotypes with high soil P 'mining' capacity (e.g. greater carboxylate and phosphatase secretion) to mobilise P fixed in the soil can also enhance P acquisition. Phosphatase secretion is one of the important adaptation strategies for P-efficient plants, which increases the hydrolysis of soil organic P to enhance soil P acquisition (Mehra et al., 2017). Efforts to develop P-efficient plants which display such traits, through breeding or genetic modification, commonly select for root morphological and P-mining traits. The mechanisms by which selected crops can enhance the release of P fixed to soil surfaces and improve crop P uptake through modifying rhizosphere properties (e.g. root system architecture and structure, phosphate transporters, key transcription factors, organic acid biosynthesis, and phosphatase secretion) should be the focus of selective breeding practices (Trolove et al., 2003; Mehra et al., 2017).

Advances in bio-fertiliser technologies (microbial biotechnologies) can contribute to the use efficiency of residual phosphorus. Many agricultural soils contain sufficient P stores to buffer food security threats for decades (Stutter et al., 2012; Menezes-Blackburn et al., 2018), although they are not immediately available for plant uptake. Bio-fertilisers include inoculants containing arbuscular mycorrhizal fungi (AMF) (Babana and Antoun, 2006), plant growth-promoting rhizobacteria (Richardson et al., 2009), and P-solubilising microorganisms (Jones and Oburger, 2011; Adhya et al., 2015; Mukhongo et al., 2017). They work to increase the turnover of P in 'plant unavailable pools' to slow the net accumulation of residual P that occurs when P-sorbing soils are fertilised. However, soil P 'mining' strategies to enhance the desorption, solubilisation or mineralisation of non-plant available P pools (Figure 4.3) are not sustainable in the long term (Richardson et al., 2011). Overuse of chemical and organic P fertilisers may suppress the functional activities of P solubilising microorganisms and AMF (Olander and Vitousek, 2000; Wang and Lambers, 2020). Thus, to ensure all P pools are sufficiently utilised, P application rates (through a combination of both organic and inorganic fertilisers) should be optimised to ensure that crop demands and the functional role of microorganisms are balanced over months to years (George et al., 2018).

Solution 4.3: Optimise animal diets to lower phosphorus excretion and improve manure management

Optimising the diets of animals in intensive farming operations to match growth requirements, and supplementing monogastric animals with phytase enzymes can reduce phosphorus excretion. Governments should provide guidance on recommended dietary phosphorus allowance for livestock based on current scientific knowledge.

Demand for animal products is increasing globally. Strategies to reduce consumption of animal products with high P footprints, and maintain healthy diets, are discussed in Chapter 8. However, multiple opportunities exist to reduce the P required to produce animal products by improving PUE in livestock production. Nutritional strategies to lower P excreted in waste streams and efficient management of manures represent key opportunities to make global improvements to P sustainability in the livestock sector.

Optimising animal diets to match growth requirements may help reduce the amount of P lost in animal manures (Wu et al., 2001; Nahm, 2002; Casartelli et al., 2005; Arriaga et al., 2009; Dersjant-Li et al., 2015). For example, Zhang et al. (2016) showed that reducing dietary P from 0.42% to 0.26% did not negatively affect growth or milk production in dairy cows, but did reduce faecal and urine P concentration by 35% and 69%, respectively. Similar studies have shown that modifying diet ingredients and composition to meet P and other nutrient requirements of the animal at different growth stages (phase feeding) (Han et al., 2001; Dao and Schwartz, 2011) can reduce dietary P excretion in cattle (Zhang et al., 2016; Guo et al., 2019), poultry and swine (Lu et al., 2017), aquaculture (Naylor et al., 2009) and horses (Saastamoinen et al., 2020), without affecting animal health or performance. Balancing P and other nutrients in diets as a front-end nutrient management approach has the advantage of saving producers' money in feed costs and lowering P surplus on farms, subsequently reducing potential environmental losses (Knowlton et al., 2004).

Government guidance on recommended dietary P allowance for livestock should reflect current scientific knowledge. Guidance in China and the USA may not be accurate and potentially results in excess P being fed to livestock (Lu et al., 2017; Guo et al., 2019), whilst in other regions, such as SSA, guidance is lacking. Evaluation of the recommendations for protein (and thus N) and P content in livestock feed is needed across regions. Around two-thirds of the P in cereal grains and oilseed meals, which make up the bulk of monogastric diets, is organically bound in the form of phytate. Monogastric animals lack sufficient digestive enzymes to digest phytate, and therefore inorganic P is added to diets, commonly in excess, to meet the requirements of the animal (Lu et al., 2017). To reduce excess P excretion in monogastric livestock, strategies to improve the bioavailability of P in feeds and subsequent reduction in P content should be implemented across all regions, especially for poultry and swine that together provide

70% of meat production (Ritchie and Roser, 2017). Supplementing the diet of monogastric animals with phytase enzymes to make P in feed grains more digestible can reduce P excretion (Poulsen et al., 1999; Nahm, 2002; Arriaga et al., 2009; Kebreab et al., 2011). Whilst this practice is already widespread in more economically developed countries, it should also be extended to less economically developed regions. It is important to ensure that the addition of phytase supplements is accompanied by corresponding and optimal reductions in dietary phosphorus. For example, for laying hens that received low P and protein diets supplemented with amino acids and phytase, N and P excretion were reduced by around 50%, with no detrimental effects on animal performance or health (Keshavarz and Austic, 2004). Similarly, pigs fed on low P content diets supplemented with phytase, excreted 19% less P than those consuming standard amounts of P in their diets, with no change to growth or animal performance (Kebreab et al., 2011). The use of phytase enzymes has allowed the poultry industry in the USA to make significant reductions in P concentrations in poultry feeds (Dou, 2003; Maguire et al., 2005; Steén , 2006). Long et al. (2017) showed that the addition of phytase to cattle diets had little effect on P absorption or retention by the animals. Whilst the use of dietary phytases can increase P digestibility in monogastric animals, the practice increases water solubility of the P excreted in manures and hence can increase the risk of P losses from land receiving manure applications (Dao and Schwartz, 2011).

Solution 4.4: Increase phosphorus recycling from manures and residue streams

Globally, recycling of treated animal manures and residues and the use of recycled fertilisers should be increased, with corresponding reductions in mineral fertiliser use. Integrating arable and livestock systems can help to reduce costs associated with transporting phosphorus-rich animal manures and residues to crops. In some cases, education, extension services and investment in infrastructure and technology are needed to support stakeholders and make phosphorus recycling more efficient.

Diet optimisation in livestock should be accompanied by manure management to optimise P recycling and minimise P loss to the environment. Manure P management may involve measures at the field scale, like the adjustment of stocking density, rotational grazing, and keeping animals away from the edges of waterways, or managing the locations of drinking water and shade to reduce the occurrence of manure hotspots within fields (Sims and Maguire, 2005; Haan et al., 2006; Webber et al., 2010; Dao and Schwartz, 2011). Where manures are to be collected, animal housing can be designed to aid collection and avoid losses. Manure storage containers should have robust construction to avoid leakage during long-term storage, with regular inspections to ensure security, and also be large enough to handle manure volumes so that application to frozen fields in winter is prevented. Furthermore, the impact storage can have on the P chemistry of manures should be considered in management strategies. For example, storage can enhance inorganic P content relative to organic P forms in manure, making it more immediately bioavailable to plants. However, increasing P solubility of manures may increase the risk of losses via convective transport and should be considered in strategies to mitigate P losses from manure applications (Dao and Schwartz, 2011). In some cases, maintaining a stable pool of organic P in manures to support the slow release of P to meet the continual needs of a plant during the growing season may be more desirable. Precision application of manures, including the placing of manures close to roots to target the crop and not the soil, can improve plant P uptake and reduce losses (McLaughlin et al., 2011; Withers et al., 2014b) (Figure 4.8). The most efficient strategies to reduce environmental impacts of manure P losses vary between different animal production systems, and particularly on the settings in which animals are raised or finished for market (Dao and Schwartz, 2011).

Despite their recognised agricultural sustainability benefits, mixed crop-livestock farms have declined in recent decades in the Northern hemisphere (Asai et al., 2018). Spatially integrating arable and livestock agricultural systems can help to reduce costs associated with transporting P-rich animal manure to crops. Whilst some farming systems rely on manure disposal contracts, local partnerships between specialist arable farms and livestock farms can support the exchange of crops, grains and manure, and coordinate land use (Lemaire et al., 2014; Martin et al., 2016) (see Chapter 6). In an assessment of 240 arable/livestock farming partnerships in Denmark, trust and reciprocal relationships enhanced through effective communication and well-functioning institutional support (e.g. local advisory services matching farmers and facilitating partnership arrangements) played pivotal roles in maintaining effective partnerships (Asai and Langer, 2014). A further study, comparing arable and livestock farming partnerships in Japan, France, the Netherlands, and the USA, demonstrated that appropriate coordination by third-party entities provided the effective financial and technical support required by partnerships (Asai et al., 2018). They argue that, in some cases, a formal legal framework for establishing crop-livestock integration may be useful to increase the credibility and permanency of partnerships.

Most, if not all, P-rich organic materials need some level of processing to reduce contaminants and pathogens to safe levels for use in food production (see Chapter 6). Many processes can be used to recover P from contaminated organic materials (Kabbe and Rinck-Pfeiffer, 2019). Whilst some P recovery processes can be expensive and provide economic barriers to recycling, the market price alone for recovered P products should not define the economic feasibility of P recovery. The economic value of the co-benefits (e.g. pollution reduction, co-production of nutrients and other critical elements and bioenergy) require better quantification to ensure economic assessments represent net societal gains. A key market for recovered P materials is an alternative raw material (i.e. to supplement PR use) for use in the mineral fertiliser industry (see Chapter 7).

Figure 4.8 Farmer applying phosphorus-rich slurry to a field using a trailing hose. Trailing hose and slurry injection techniques offer the potential to reduce dissolved phosphorus concentrations in runoff during the period immediately after slurry application.

Phosphorus lost from agricultural land but which has accumulated in aquatic ecosystems (e.g. within biomass and bed sediments) (Sharpley et al., 2013; Powers et al., 2015), may provide a limited source of P for agricultural soils. This recovery pathway may be more beneficial to the P receiving environment as an effective P reduction measures, but may also provide some level of organic P to support local agriculture. For example, recycling fish-pond sediments has been demonstrated as a source of plant nutrients with additional soil conditioning benefits (Rahman et al., 2004; Rahman and Yakupitiyage, 2006; Ihejirika et al., 2011). This may be relevant in regions such as Asia, where aquaculture is increasing at significant rates (Huang et al., 2020) (see Chapter 5). The use of such materials should be explored further, especially for supporting smallholder farms where access to inorganic P fertilisers may be limited. However, it is important to consider that lakebed sediments in some regions may be highly contaminated, for example, where mining activities are or have been, prevalent and where cyanobacteria toxin concentrations are high. Furthermore, the removal of sediments from aquatic ecosystems will itself create a damaging impact (through habitat loss) on benthic aquatic organisms.

To meet regulations and fulfil ‘organic food’ certifications from most international organic food associations, organic farmers cannot use ‘conventional’ mineral P fertilisers (e.g. diammonium phosphate, monoammonium phosphate, single superphosphate, and triple superphosphate) (Stabenau et al., 2018). To avoid depleting soil P levels organic farmers must rely on recycled P sources. For organic farms without livestock or access to sufficient manures, fertilisers made with P recovered from organic residues (e.g. food wastes, seaweeds, biochar, products or by-products of animal origin) can be used. A full list of fertilisers, soil conditioners and nutrients permitted for use in organic farming systems in the EU is provided in Annex 1 of European Commission (2008), although products or by-products of animal origin (including blood, bone, and fish meal) must not be applied to edible parts of the crop. Ground PR can also be applied to soils, and is allowed in organic production systems, but is not an effective source of P in most soils, except those with low pH (Nesme et al., 2012).

Measures to increase the recycling of P-rich organic materials and recovered P products are discussed in detail in Chapters 6 and 7.

Solution 4.5: Develop integrated policies and phosphorus use efficiency targets across scales

An integrated approach is essential to increase sustainable phosphorus use in the agricultural sector and will require actions across scales, sectors, disciplines, and regions. Targets to increase phosphorus use efficiency in agriculture and indicators to monitor improvement from farm to global scales are needed. Phosphorus budgets at the farm level are needed to develop catchment management plans that scale phosphorus use efficiency assessments to national, regional, and global scales. We must maximise synergies with other nutrients and ensure that policies are adaptive.

Targets to increase PUE in agriculture, and indicators to monitor improvement, are needed at national, regional, and global scales. Policymakers can help address this need by developing and implementing enabling policies (McDowell et al., 2016) to support the delivery of PUE targets. Enabling policies could promote, for example:

- soil testing and plant tissue analysis to inform P fertiliser use recommendations (Masso et al., 2017; Blackwell et al., 2019);
- optimisation of P budgets in soils to appropriately match farming systems and soil types (Ohm et al., 2017; Zhou et al., 2017; Lun et al., 2018);

- improving the formulation of animal feeds to avoid excess losses in manures (Knowlton et al., 2004; Kleinman et al., 2019); and
- the implementation of safe threshold limits for cadmium and harmful contaminants in mineral and recycled P fertilisers.

Policies could also define national targets for P recycling, PUE and P losses. Enforcement, or development and implementation of supportive policies are required to create an enabling environment to make recommended options for improving PUE economically viable (Withers et al., 2014b, 2015). In some cases, financial instruments such as subsidies, tax incentives, and support will be required for farmers to adopt sustainable measures. In some regions, infrastructure development will be necessary to support measures to increase PUE, for example, where collection services and transport networks for P-rich organic materials are currently insufficient.

As highlighted in the solutions above, an integrated approach to improve full chain PUE and reduce losses throughout the food production chain is needed. A multiple stakeholder approach will, therefore, be critical. Whilst changing farming behaviours is a key requirement, farmers cannot make changes without supporting actions also being implemented throughout the food production and consumption chain (i.e. the network of stakeholders involved in growing, processing, and selling the food that consumers eat). Stakeholders in this chain must be collectively engaged and their roles in delivering PUE gains supported, including, farmer organisations, extension services, private and public sector bodies, policymakers, and the scientific community.

Stakeholders must be appropriately consulted on the development of national strategies to ensure that they reflect local needs and available resources. For example, the central role of farmers organisations in agri-environmental schemes in Canada significantly enhanced good P management in the country's agricultural sector and increased acceptance of the recommended solutions (Robinson, 2006). Opportunities to adopt more alternative and more P sustainable farming behaviours will differ between regions, countries, farm types and individual farmers. It will be important to ensure that there is a common understanding of the barriers (physical, social, cultural, economic and political) to good P management and the options to overcome them (Scholz et al., 2014). For example, the social and cultural factors influencing water pollution mitigation behaviours within the farming community must be understood, so that farmer engagement is sustained over the timescales needed to deliver lasting reductions in P losses (Inman et al., 2018).

To ensure crop yields, integrated management of N, P and other nutrients is required (Kanter and Brownlie, 2019). Whilst 'traditional' application of manures to fields provides most of the nutrients needed for crop growth (N, P, and micronutrients), the relative proportion of nutrients rarely matches the needs of the crop. This can result in the over-application of some nutrients, especially when manures are applied primarily as a source of N, with little consideration of soil P accumulation that can result from repeated manure applications (Shober and Sims, 2003; Sims and Maguire, 2005; Bouwman et al., 2017).

In addition to tackling structural and cultural barriers within the farming sector, P sustainability strategies should also take account of the pressures of climate change and population and economic change. This will require a clear understanding of the combined effects of climate drivers, source management, and hydrological and chemical controls in the landscape, and how they impact P transfer from soils to groundwater and surface water. In an assessment of the impact of projected climate change on future phosphorus transfers in three UK catchments, Ockenden et al. (2017) showed winter P transfers from land to waters would increase by 30% by the 2050s, and that limiting these losses would only be possible with large-scale agricultural changes (e.g. 20–80% reduction in P inputs). Since such reductions may not be compatible with future demands for agricultural productivity, policymakers will need to reassess priorities, as outlined in Doody et al. (2016). There is a critical need to increase our understanding of the effects of climate change on PUE to underpin the development of long-term mitigation options (Al-Kaisi et al., 2013). Integrated climate-hydro-chemical indicators will be useful for shaping future P policy, to ensure they are sufficient to optimise PUE, whilst mitigating P losses from agricultural soils to water. However, it is clear that to prevent overestimation or underestimation of fertiliser requirement over time, the management of P in agricultural systems should be a dynamic process, underpinned by an adaptive policy approach (Syers et al., 2008; Blackwell et al., 2019).

Fortunately, measures that reduce P losses and improve PUE in crop production are 'win-win'. Whilst they aim to increase crop production by reducing the need for external P application, this can improve food security, reduce P transfer to waterbodies and associated eutrophication, and in some regions reduce costs wasted on the application of P fertilisers that are not needed. In this way, addressing P sustainability in agriculture delivers on multiple United Nations Sustainable Development Goals (SDGs) including, poverty alleviation SDG 1 - Zero Hunger, SDG 2 - Clean Water and Sanitation, SDG 6 - Responsible Consumption and Production, SDG 12 - Life Below Water, SDG 14 - Life on Land (SDG 15).

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