06

Opportunities to recycle phosphorus-rich organic materials

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Left: Cattle eating hay on a farm in the New Forest District, UK. Manure is a valuable phosphorus resource; its use as an organic fertiliser should be optimised and carefully managed to avoid phosphorus losses. Photographed by Annie Spratt on www.unsplash.com - www.anniespratt.com

Recycling phosphorus-rich organic residues and manures is critical for phosphorus sustainability and a transition to a more circular economy for phosphorus. Beyond agronomic benefits, the win-wins are numerous, with benefits to society, environment, economy, and business growth. However, to significantly increase phosphorus recycling, education, awareness-raising, investment in technology and infrastructure, and policy support are urgently needed.

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Challenge 6.1: Organic wastes and residues are often treated as pollutants and not nutrient resources

Organic materials are often managed as pollution rather than as a valuable nutrient resource. Consequently, improvements in the management of phosphorus-rich organic materials are necessary including collection and storage, processing, and application practices. Farmers and stakeholders may reject recycling some organic materials as fertilisers because of negative perceptions over the safety of their use in food production; these concerns must be overcome.

Challenge 6.2: Manure and waste production is often 'decoupled' from croplands where it can be recycled

In many regions, the distances between the production of phosphorus-rich organic materials and arable land are increasing, driven by the expansion of specialised and intensive farming, urbanisation, and globalised trade. This can make transporting such materials to areas where they can be recycled prohibitively expensive. Decoupling of livestock and arable farming systems is particularly problematic for farmers producing organic foods and feeds. This is because 'conventional' mineral phosphorus fertilisers, and in some cases manures from confined animal feeding operations, cannot be used to fertilise organic crops.

Challenge 6.3: The reliability of phosphorus-rich organic materials is often lower than mineral fertilisers

The concentrations of phosphorus in organic materials are variable, not easy to determine quickly and lower than mineral phosphorus fertilisers, representing a challenge for farm-scale nutrient management. The bioavailability of phosphorus in organic materials varies and influences their performance as fertilisers, and can be affected by soil type, pH, and crop breed. The bulky nature of many organic materials can make them difficult to spread consistently, affecting their reliability as a fertiliser.

Challenge 6.4: Some phosphorus-rich organic materials can contain contaminants

Pathogens, hormones, antibiotics, potentially toxic elements, and microplastics can be present in some phosphorus-rich organic materials. It is important to ensure contaminants are removed, destroyed or concentrations reduced to safe levels in any phosphorus-rich organic materials to be used as fertilisers. In some cases, contaminants can accumulate in soils and may pose a risk to human and animal health and environmental quality.

Challenge 6.5: Policy, infrastructure, and financial support are lacking for phosphorus recycling

There is a lack of coordinated policy and regulation to support an increase in the recycling of phosphorus-rich organic materials. In some regions, there is little economic incentive for farmers to switch from mineral phosphorus fertiliser to phosphorus-rich organic materials. Some farmers can face legal and certification barriers stopping them from recycling certain phosphorus-rich organic materials.

Solution 6.1: Treat waste streams as valuable nutrient resources

A paradigm shift in how we view waste streams is needed; from pollutant to valued nutrient resource. Key actions in delivering this shift include raising awareness of the costs of phosphorus losses and benefits of phosphorus recycling, providing education and extension services to encourage stakeholders to recycle phosphorus, and mobilising investment in infrastructure and technology to make phosphorus recycling safe, easy, and efficient.

Solution 6.2: Optimise the spatial integration of arable and livestock agricultural systems

Landscape planning can integrate arable and livestock farming to maximise nutrient recycling. Whilst efforts should be made to ensure animal densities in livestock farming do not exceed nutrient needs, some farming systems must rely on disposal/utilisation contracts. Arable-livestock farming partnerships can support the exchange of crops, grains, and manures, and coordinate land-use to support more regionally closed feed-manure loops.

Solution 6.3: Utilise available technology and tools and provide education

The reliability of phosphorus-rich organic materials as fertilisers can be improved by processing to improve fertiliser quality, and developing better systems to help farmers assess the phosphorus content and phosphorus bioavailability of the materials. Furthermore, farmers can be better supported to optimise the application of recycled phosphorus products and other nutrients in order to maximise phosphorus uptake by plants. However, critical to this is a sufficient understanding of farm- and local-scale nutrient budgets.

Solution 6.4: Process organic materials appropriately and provide safety certification schemes

Most phosphorus-rich organic materials need some processing to reduce contaminants and pathogens to safe levels for use in food production. Reducing livestock dietary intake of potentially toxic elements and imposing strict limits on the non-therapeutic use of antibiotics in livestock, will reduce levels of these contaminants in manure and biosolids. Assurance that fertiliser products derived from phosphorus-rich organic materials are safe for their intended use should be provided to end-users.

Solution 6.5: Develop policies, regulations, and financial instruments that support phosphorus recycling

Improved coordination between relevant government bodies and relevant stakeholders is required to develop coherent, holistic policies and create markets for recovered phosphorus fertiliser. Investment in infrastructure and technologies supported by cross-sectorial innovation, co-creation and sharing of knowledge can help to make phosphorus recycling simple and efficient. The economic benefits for society of recycling phosphorus need to be better quantified and used to encourage stakeholders to recycle phosphorus more efficiently. The value of recovering phosphorus can be maximised by selecting methods to process organic materials that produce additional co-benefits.

6.1 Introduction

In the natural biogeochemical phosphorus (P) cycle, P is released into ecosystems by the weathering of phosphate rock (PR). Rivers and food webs slowly cycle P through landscapes until ultimately it is deposited into the oceans (Huang et al., 2020). Over millions of years, new PR deposits are created in ocean sediments. By mining PR, we have accelerated global P mobilisation fourfold (Falkowski et al., 2000). Currently, the main input of P to the anthropogenic P cycle is P mined from phosphate rock. Around 85% of mined P is used for fertilisers applied to soils to grow crops, which are either consumed directly by humans or used to feed livestock (de Boer et al., 2019). Significant P losses occur throughout the food production system (see Chapters 4, 5 and 8). However, much of this lost P remains on land, and can be considered misplaced (Dawson and Hilton, 2011), and therefore potentially recyclable within the agricultural system (see Chapter 4). In this chapter, we discuss potential opportunities to recycle this misplaced P, valuing it as a resource as opposed to waste.

6.1.1 Defining phosphorus recycling, phosphorus losses and the circular economy.

In the literature, the terms P recycling, P recovery and P reuse have been blurred. A common and general definition of recycling is a process that converts waste materials into new materials. In this report, we define P recycling as the use of P from residue streams (e.g. manure, biosolids, food wastes) in the production of food (e.g. crops and vegetables) and non-food agricultural products (e.g. fibre and timber). This definition highlights a key goal of P recycling, which is to offset demand for P from mined sources. The most direct method of P recycling is the application of manures and biosolids to cropland, in which P is returned to soils, where plants can assimilate it back into agricultural products.

In some cases, P must be 'recovered' from wastes before they can be recycled safely and effectively. Phosphorus recovery refers to processes used to isolate high-quality P from organic matter into raw materials that can be used to make recovered P fertilisers, or materials for use in the chemical industries. Whilst the use of P recovered from waste materials to produce fertilisers could be considered recycling (i.e. a waste material converted into a new material), in this report, we consider it only a stage in the P recycling process, as it is yet to be used in agriculture. This definition is used to distinguish P recycling and P recovery and is illustrated in Figure 6.1. The only exception to our definition of P recycling could be when recovered P is used to produce raw materials not related to soil fertilisation (e.g. to make food additives). However, to our knowledge, this is not done at large scale (and thus this pathway is not included in Figure 6.1). It is important to note that policies that enforce P recovery from waste streams, as in some EU countries, may not enforce P recycling. In this chapter, we focus on the use of P-rich organic materials as a source of nutrients to fertilise agricultural soils. Processes to recover P from waste streams to produce raw materials that can then be used to make customised recovered P fertilisers, and other specialised products, are discussed in Chapter 7.

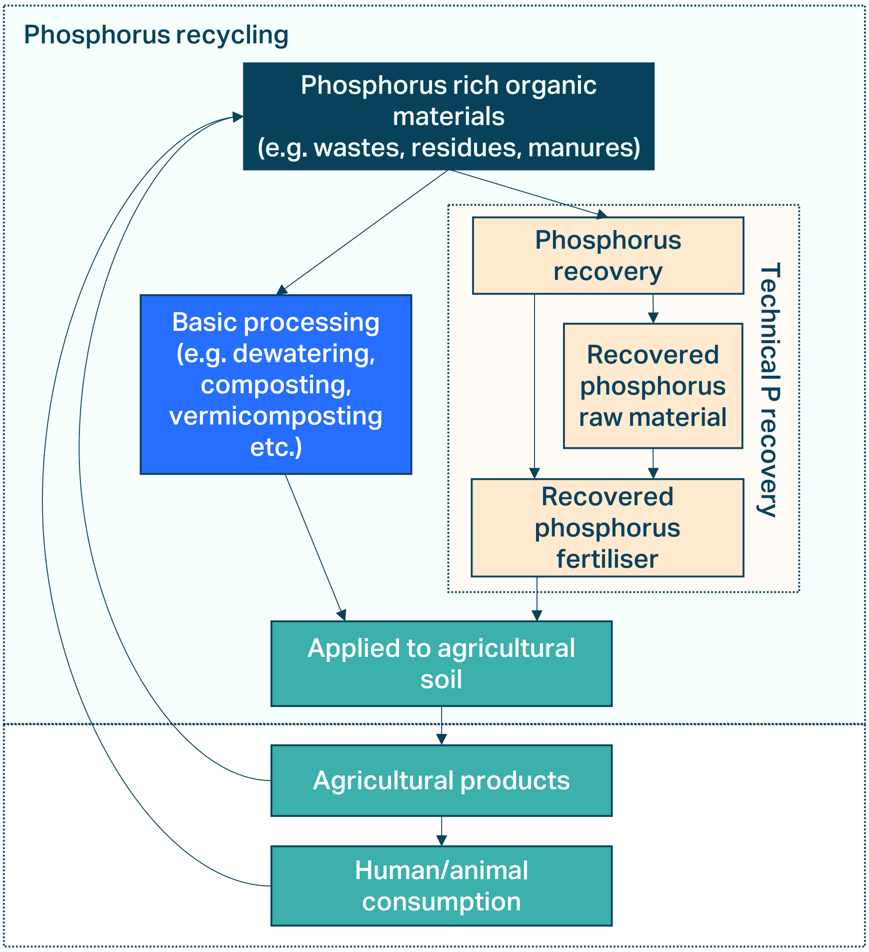

Figure 6.1 Conceptual diagram to illustrate the boundaries used in this report to distinguish between phosphorus (P) recycling and P recovery, demonstrating the circularity of P once it is in the agricultural/food system. Recycling P from organic wastes without technical P recovery is discussed in this chapter, whilst technical P recovery (which can be considered a stage in recycling P rich organic wastes) is examined in Chapter 7.

The term 'reuse' is commonly used alongside P recovery (i.e. 'P recovery and reuse'). Reuse, in the context of recycling, has been defined as the transfer of products to new owners (Fortuna and Diyamandoglu, 2017). In this report, we use the term P recycling but not P reuse, though acknowledge that P reuse is used in the literature (Cordell et al., 2011; Karunanithi et al., 2015; Sun et al., 2018).

The term 'P losses' are commonly used to describe P inputs to anthropogenic systems that do not contribute to productive output and underpin P use efficiency (PUE) calculations (see Chapter 4). A more accurate description of P losses is perhaps P dissipation (Dawson and Hilton, 2011). The extent to which P is dissipated, to some degree, defines how easy it is to recycle. For example, P that enters the oceans can be considered truly 'lost' (Dawson and Hilton, 2011), when compared with P in human wastes (i.e. faeces and urine), which can be more easily captured, recovered through processing (where necessary), and recycled. Phosphorus losses occur at all stages of the food production and consumption chain, enabling a circular economy approach to be implemented when framing the opportunities around recycling and recovery (Geissler et al., 2018) (Figure 6.2). A circular economy can be considered an alternative approach to a traditional linear economy (i.e. make, use, dispose of) in which resources are kept in use for as long as possible, then recovered to regenerate new products and materials. Whilst various definitions of a circular economy exist, and have been discussed in the literature (Kirchherr et al., 2017), a central aim shared amongst definitions is to decouple economic growth from the consumption of finite resources. The recycling of P to reduce consumption of finite PR reserves, is, therefore, a key driver in the transition towards a more circular P economy (Geissler et al., 2018) (Figure 6.1), as set out generally by regional (European Commission, 2015) and international initiatives (UNEP, 2017).

6.2 Sources, types, and fates of phosphorus-rich organic materials

There is abundant P present in organic residue streams that can be used to improve soil fertility to optimise crop yields (Figure 6.2). Current P-rich residue streams have been identified to support P recycling including:

- crop residues (Jat et al., 2015; Espinosa et al., 2017);
- manures and slurry (Komiyama et al., 2014; Omara et al., 2017; Kumaragamage and Akinremi, 2018);
- food processing residues, including from aquaculture (Hamilton et al., 2017);
- abattoir residues (especially bone meal which is high in P) (Darch et al., 2019);
- domestic food wastes (Nakakubo et al., 2012);
- sewage derived biosolids (Deeks et al., 2013; Pawlett et al., 2015; Antille et al., 2017);
- wastewaters (Egle et al., 2016; Cieślik and Konieczka, 2017).

In most cases, farmers can apply crop residues and urine to croplands directly without the need for potentially costly recovery processing. However, P-rich organic materials may require processing to remove contaminants, concentrate nutrients, reduce volumes for transport, and improve P bioavailability (Figure 6.2). Direct manure spreading may be environmentally acceptable for some small-scale organic farming systems. However, Font-Palma (2019) argued that the direct spreading of cattle manure onto land carries the risk of potential release of greenhouse gases, odour, contaminants and pathogens into the environment and that in the future biological or thermochemical conversion technologies should be more widely applied to reduce these undesirable effects.

Phosphorus in waste streams, soils or waters can be extracted through biological assimilation into microorganisms, plants and animals to support recycling (Guterstam, 1996; Gifford et al., 2007; Naylor et al., 2009; Spångberg et al., 2013). This has been done as part of environmental restoration efforts (Delorme et al., 2000; Novak and Chan, 2002) and to clean waste streams, and in some cases, the extraction media (e.g. algae) can then be applied to soils as a P source. However, in such cases, the risk of negative human health effects associated with the ingestion of toxins (e.g. toxin-producing cyanobacteria, Chapter 5) should be carefully assessed.

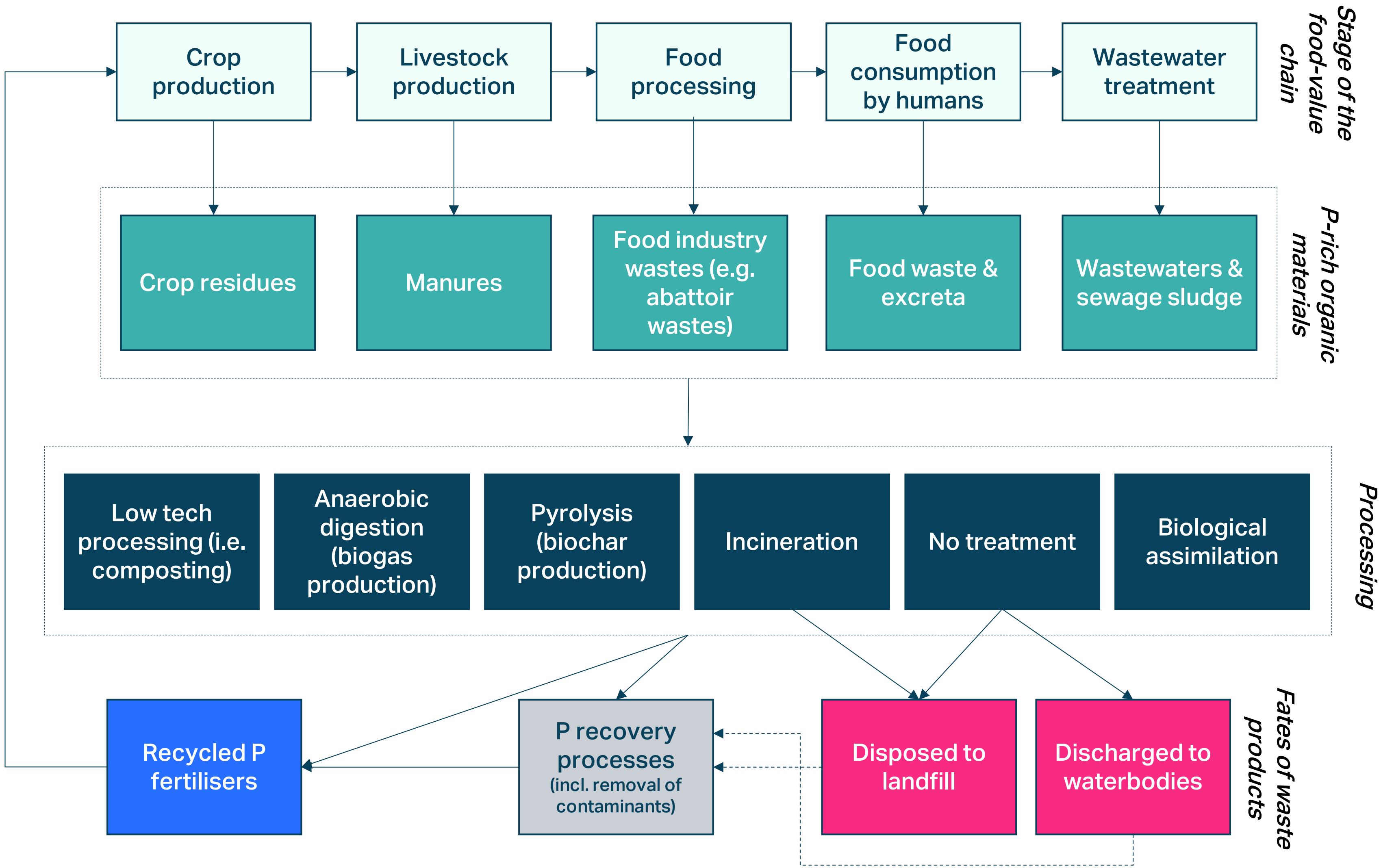

Figure 6.2 Conceptual visualisation of the food value chain showing the direction of phosphorus (P) flows (solid arrows = common flows; dotted arrows = uncommon flows), stage of the food value chain (light blue), losses of P-rich organic materials (green), common methods of processing P-rich organic materials (dark blue), the pathway to recycled P fertilisers (which includes recovered P fertilisers and treated P-rich organic wastes) (blue), and the cycle back to agricultural soils with or without recovery processing (grey). Some organic wastes are lost to the environment (pink) without treatment or following incineration. The products of P recovery are mostly used to make recovered P fertilisers, although recovered P may be used for products that are not applied directly to agricultural soils; this P flow is minimal and is not shown.

Phosphorus-rich organic materials can also be used to make bioenergy (Huygens and Saveyn, 2018), biogas (Lansing et al., 2010; Insam et al., 2015), and biochar (Lehmann et al., 2006; Atkinson et al., 2010; Blackwell et al., 2015; Trazzi et al., 2016), producing P-rich residues as a by-product. Biochar and biogas residues can be applied to soils directly, and are effective slow-release P fertilisers under certain conditions (Tsachidou et al., 2019; Glaser and Lehr, 2019). Further information on common treatment processes of P-rich organic materials is summarised in Table 6.1. Some of these processes can be further combined with other physical, chemical, or biological treatment options to create more specific fertiliser products (see Chapter 7), especially in the case of human excreta related waste streams (Harder et al., 2019).

6.2.1 Mineral phosphorus inputs accommodate high phosphorus losses

The recycling of organic materials from residue streams is sub-optimal, and the need to 'close the P loop' is widely acknowledged (Elser and Bennett, 2011; Bateman et al., 2011; Cordell and White, 2014; Scholz and Wellmer, 2018; Withers et al., 2018).

If all P losses could be recycled (i.e. closing the anthropogenic P cycle), additional P inputs would only be needed to support population growth and replace the ~1.0 Mt P year^{-1} lost in dead human bodies (which tends not to be recycled) (Dawson and Hilton, 2011). This estimate is in terms of mineral P and is based on an average person containing 780 g P (CRC, 2005). However, in 2020, 21 Mt of mineral P was added to the anthropogenic P cycle (Jasinski, 2021), 85% of which was used in mineral fertilisers (de Boer et al., 2019) (see Chapter 2). Increasing the relative proportion of recycled versus mineral P (i.e. PR derived P) is essential to redress the global anthropogenic flow of P, which was identified as having passed its planetary boundary a decade ago (Carpenter and Bennett, 2011). During the same decade, global consumption of mineral P fertiliser increased and is projected to continue for at least the immediate future (IFA, 2020)[i]. Without implementing sustainable P management to reduce P losses and increase P recycling, environmental damage will continue to increase with potentially irreversible consequences (Steffen et al., 2015; Rockström et al., 2020; see also Chapter 5).

[i]The International Fertiliser Association forecasts an 0.8% annual increase in consumption of P_2O_5 in fertiliser in the short term, based on a three-year average of 2017, 2018 and 2019, and the end of its outlook forecast in 2024 (IFA, 2020).

Table 6.1 Description of key phosphorus (P) treatment processes for P-rich organic materials including post-treatment products, the benefits and disadvantages of the processes, and key references providing further information.

Description of treatment process	Organic materials commonly treated	Post-treatment (recycled) product details	Benefits / disadvantages	Reference
Composting. An aerobic process that uses biological degradation by mesophilic and thermophilic microorganisms.	Crop residues, animal manures, human faeces, food wastes, aquatic macrophytes, algae, biosolids.	Produces humus-like materials containing both nutrients and microorganisms. Heat eliminates most pathogens. However, depending on the waste source, herbicides, plastics, metals, pharmaceuticals, persistent pathogens, and potentially toxic elements may persist.	Agronomical benefits, such as slow-release nutrients and the addition of organic carbon to soils.	Goyal et al. (2005) Martínez-Blanco et al. (2013)
Vermicomposting. Accelerated bio-oxidation and stabilisation of organic materials involving earthworms.	Human wastes, sewage sludge, paper waste, brewery waste, animal manure, food processing wastes, industrial wastes.	Produces humus-like materials containing both nutrient and microorganisms, but richer in bioavailable P than bio-wastes treated with mineralisation processes. Like composting, heat eliminates most pathogens, although, depending on the source, contaminants need monitoring.	Faster than composting, with a more diverse population of microorganisms.	Domínguez et al. (2004) Yadav et al. (2010) Moya et al. (2019b)
Anaerobic digestion. The degradation of organic materials to methane, by microbes in a digester, as used in biogas production.	Livestock manure, crop residues, food wastes, sewage sludge.	Biogas residues can be used as a fertiliser. Concerns over nutrient mobility (i.e. P leaching) have been raised. Composting biogas residues with carbon-rich materials or applying them to aquaculture to grow animal feed (e.g. duckweed) may help mitigate this. Phosphorus in biogas residues may not be immediately plant bioavailable.	Harnessing captured methane for energy.	Lansing et al. (2010) Insam et al. (2015)

Description of treatment process	Organic materials commonly treated	Post-treatment (recycled) product details	Benefits / disadvantages	Reference
Pyrolysis/Biochar production. The solid product of organic material pyrolysis; a process by which organic material is heated to temperatures of at least 150 °C under oxygen-limited conditions.	All organic materials.	There is potential for biochar to serve two different purposes: i) as a sorbent for removing P from aqueous solutions, and ii) to enhance the plant availability of P within agricultural soils. The source of biochar (e.g. manure, sewage sludge, crop residues) will influence its chemical and physical properties.	Facilitation of carbon storage, increased soil aeration, enhanced water storage, and growth promotion of microorganisms and mycorrhizal fungi that benefit plant nutrition.	Lehmann et al. (2006) Atkinson et al. (2010) Blackwell et al. (2015) Trazzi et al. (2016) Shepherd et al. (2016)
Incineration. Incineration of organic materials, either with or without energy recovery, is a means to destroy pathogens and organic matter and typically results in a 90% reduction in volume and 60% reduction in weight	All organic manures, commonly sewages and manure.	Incineration ashes are taken to landfill or used in construction materials. Processes to recover P from ashes of organic residues are available, but not mainstream.	Incineration safely removes contaminants and offers the opportunity to recover combustion heat. Energy-intensive and produces carbon dioxide (CO_2). May also require chemical scrubbers to remove gaseous pollutants such as nitrogen oxides (NOx).	Chandler et al. (1997) Xu et al. (2012) Wang et al. (2018)
No treatment/direct use of organic materials. None or minimal processing (e.g. dewatering of biosolids and manures to reduce transport volumes, whilst retaining nutrient content).	Animal manures and slurry, crop residues, food wastes, human urine	Livestock manures, human urine and crop residues and food wastes can be directly ploughed into soils. However, careful manure management is required (i.e. method of application, the timing of delivery) to avoid over fertilisation and nutrient losses. Manure contaminants must be monitored to avoid transmission to foods. Manures are commonly applied directly to fishponds in aquaculture in parts of Asia.	Improves soil health and function. Use has low technology requirements, with little processing costs. However, in some cases, costs can be prohibitively expensive (see 'decoupling' below).	Wohlfarth and Schroeder (1979) Corbala-Robles et al. (2018)

Description of treatment process	Organic materials commonly treated	Post-treatment (recycled) product details	Benefits / disadvantages	Reference
Enhanced biological phosphorus removal (EBPR). Heterotrophic bacteria (polyphosphate-accumulating organisms (PAO)) are cultured within activated sewage sludge under anaerobic conditions. The PAO are then passed into aerobic conditions, where they can take up polyphosphates in excess of that required for normal biomass growth, known as 'luxury uptake'.	Sewage sludge, animal manures and slurry.	The sludge can be dried and applied directly to the soil as fertiliser (with high transport and application costs and need for nitrogen amendment) or P can be recovered from the sludge by biological or thermo-chemical methods, including struvite recovery.	Treatments using EPBR do not require chemicals, although, the configuration of the plant can be complex, requiring anaerobic and aerobic zones. If P recovery is not implemented, anaerobic digestion of EBPR sludge can result in undesired struvite precipitation in the digester or the equipment around the digester. Alternative configurations and novel systems are available to solve this problem.	Kuba et al. (1993) Morse et al. (1998) Guisasola et al. (2019)
Assimilation by macrophytes, algae and trees. Harvesting P in the biomass of macrophytes (e.g. water hyacinth, duckweed and western waterweed) and algae in ecological wastewater treatment systems.	Livestock manure and slurry, urine, domestic wastewater, biogas residues.	Assimilation of P into macrophytes or algae grown in P-rich wastewaters or eutrophic waters can be harvested and recycled (e.g. as. feed, fertiliser, structural materials). Floating mats of plants made of buoyant materials have been used to assimilate P into plant biomass in eutrophic lakes. This can often be a slow process of P recovery.	Benefits of plant products (e.g. feed, habitat), and photosynthetic conversion of carbon dioxide to biomass. Some organisms may bioaccumulate contaminants, which may be a disadvantage if recycled.	Mulbry et al. (2005) Shilton et al. (2012) Stabenau et al. (2018)

Description of treatment process	Organic materials commonly treated	Post-treatment (recycled) product details	Benefits / disadvantages	Reference
Phytoextraction. Using plants to extract P from land, such as corn silage and Indian mustard seed - i.e. 'P-hyperaccumulators'.	P-rich soils.	Aims to increase the amount of P mined from soils and the P content of the resulting above-ground biomass. Requires knowledge of soil properties and P forms, as well as plant physiology. Certain tree species can be used to extract P from deeper soils (i.e. `nutrient pumping').	Carbon sequestration and ecosystem benefits of plants and plant products (i.e. feed/food/habitat). Some organisms may bioaccumulate contaminants, which may be a disadvantage if recycled.	Delorme et al. (2000) Novak and Chan. (2002)
Zooextraction. Using animals to extract P from the environment.	P-rich waters or soils.	Ideal species for zooextraction of nutrients from aquatic systems must have a capacity to accumulate nutrients, with resistance to toxicity, have rapid growth rate, and be non-invasive, easily cultured and relatively sedentary, or natural and harvestable. In Sweden, mussel farms have been used to harvest P from seawaters. Terrestrial and aquatic vertebrates can be used to recover environmental P in bones (P in bone can have low P bioavailability without further treatment).	Production of animal products (i.e. feed/food). Some organisms may bioaccumulate contaminants, which may be a disadvantage if recycled.	Gifford et al. (2007) Spångberg et al. (2013)
Aquaponics. Integrated animal husbandry-fish farming systems and sewage treatment fish farming systems.	Fish excreta, residues, feed scraps, manures and crop by-products.	Aquaponics are systems consisting of hydroponics and aquaculture elements where fish are farmed, and water enriched with nutrients from fish excreta is used to stimulate plant growth. However, harvests of P in fish and plant products can be very modest.	Replaces the use of commercial fish feed pellets which carry environmental impacts (e.g. overexploitation of forage fisheries). Potential accumulation of contaminants.	Guterstam (1996) Naylor et al. (2009)

6.2.2 Recycling phosphorus-rich organic materials deliver multiple benefits

Assessments have identified that recycling P from waste streams can significantly support a reduction in mineral P fertiliser requirements, whilst increasing soil fertility, in the UK (Bateman et al., 2011), Sweden (Akram et al., 2019; Lorick et al., 2021), the EU (van Dijk et al., 2016), India (Naresh et al., 2018), Pakistan (Akram et al., 2018), the USA (Metson et al., 2016), China (Bai et al., 2016a) and globally (Menzi et al., 2010). Improvement in soil fertility and function due to the application of organic materials can include an increase in nitrogen, micronutrients, organic carbon and water retention (Schröder, 2005; Lashermes et al., 2009; Diacono and Montemurro, 2010). Furthermore, converting waste products into useable products can support the growth of new businesses and contributes to the circular economy (Kabbe, 2019).

Alongside the agronomic benefits of recycling P-rich organic materials, the wins-wins of recycling P-rich organic materials are numerous, with benefits to society, the environment, and the economy. Organic fertilisers, when available in sufficient quantities, provide beneficial soil organic matter that improves soil health, fertility, structure, and water retention capacity, and adds micronutrients essential for plant growth and for boosting the nutritional value of crops. Mineral and organic fertilisers can play complementary roles, with mineral fertilisers supplementing the nutrients provided by organic fertilisers with concentrated, consistent nutrients that are immediately available for plant uptake. Greater recycling of P-rich organic materials will help to deliver on the objectives of multiple United Nations Sustainable Development Goals (SDGs) including SDG 1- Poverty Alleviation, SDG 2 - Zero Hunger, SDG 6 - Clean Water and Sanitation, SDG 12 - Responsible Consumption, SDG 12 - Life on Water, SDG 15 - Life on Land. In the following section, we discuss the challenges and solutions for recycling phosphorus-rich wastes and manures.

6.3 Challenges

Challenge 6.1: Organic wastes and residues are often treated as pollutants and not nutrient resources

Organic materials are often managed as pollution rather than as a valuable nutrient resource. Consequently, improvements in the management of phosphorus-rich organic materials are necessary including collection and storage, processing, and application practices. Farmers and stakeholders may reject recycling some organic materials as fertilisers because of negative perceptions over the safety of their use in food production; these concerns must be overcome.

In an assessment of global P flows in 2013, it was estimated that about 30% of the P in animal manures (equivalent to ~4 Mt P year^{-1}) and 85% of the P in human excreta and other human wastes (equivalent to ~6 Mt P year^{-1}) were not recycled (Chen and Graedel, 2016). These values vary widely between countries and assessments. National P flow assessments are often not comparable because the quality of data for 'recycling rates' or 'recovery rates' vary, as do operational definitions of 'recycling' and 'recovery' (e.g. Chowdhury et al., 2014; van Dijk et al., 2016; Rahman et al., 2019). Nevertheless, it is evident that in all regions, valuable nutrients in organic residue streams are being lost to the environment or landfill, at all stages throughout the food value chain (Figure 6.2). These losses are not only a waste of P but also a pollution risk (see Chapter 5). In some cases, significant amounts of manure and organic residues are discharged directly into waterbodies (Sattari et al., 2014; Strokal et al., 2016).

Livestock manures represent the greatest source of P-rich organic material. The amount of nutrients excreted by livestock globally is uncertain because of poor data on the intake and consumption of livestock feed (Menzi et al., 2010). The global livestock excretion rate is estimated at 16 Mt P year^{-1} (Chen and Graedel, 2016). Cattle contribute about 40% of the total livestock P excretion, whilst pigs and poultry contribute about 20% each (Menzi et al., (2010). In the EU, about 1500 Mt of animal manure is produced annually (Holm-Nielsen et al., 2009), of which 70% is recycled (van Dijk et al., 2016). However, in other regions manures are less effectively handled, such as in China, where in some catchments up to 64% of the P in manure can be discharged to rivers (Strokal et al., 2016). Similarly, high manure P losses, and low recycling rates, have been documented in countries across East and South-East Asia, and South America (Menzi et al., 2010; Teenstra et al., 2014). In many parts of the world, more sustainable manure management is hindered because it is still considered a 'waste', rather than a valuable nutrient source (Menzi et al., 2010).

Phosphorus losses from domestic and food processing residues (e.g. abattoir residues), and human excreta and other human wastes has been estimated at ~11 Mt P year^{-1} (Chen and Graedel, 2016). Recycling of the P in these waste streams is low (<20%), with significant amounts lost to the environment or discarded to landfills

(Ott and Rechberger, 2012; Chowdhury and Chakraborty, 2016; Bai et al., 2016b; Rahman et al., 2019). Currently, in the EU about 20% of municipal wastes (which include food waste and wastewater) are recycled (van Dijk et al., 2016). More than 10 Mt of biosolids (weight of dry solids) are produced annually in the EU (Laturnus et al., 2007), and this value was predicted to rise to 12.8 Mt year^{-1} by 2020 (European Commission, 2008a). Whilst food waste is also increasing, already one-third of the food produced for human consumption is lost or wasted globally, amounting to 1,300 Mt year^{-1} (FAO, 2011). Whilst collection of household food wastes for compost is common in parts of the EU (Sörme et al., 2019), in 2017 only 17% of the EU's municipal wastes were composted with the remainder ending up in mixed wastes that are landfilled or incinerated (European Environment Agency, 2020). Concentrated centres of consumption and disposal, such as restaurants, hotels, service stations, businesses, schools, universities, army barracks and hospitals, have an opportunity to collect large amounts of food waste and human excreta for recycling (Drangert, 2012). Phosphorus recycling is particularly relevant to abattoirs. Animal bone has a very high P content compared to other animal wastes and residues. For example, the P content of bovine and poultry bone is about 10% of its dry weight (Beighle et al., 1993; Hemme et al., 2005). Indeed, 85–88% of the P in vertebrates exists in the skeleton (Hua et al., 2005). However, in the EU, alone, some 4 Mt year^{-1} of animal bone biomass is produced (Someus and Pugliese, 2018), most of which is discarded to landfills (Ayllón et al., 2005; Dawson and Hilton, 2011).

Poor sanitation is allowing P-rich excreta to pollute waterbodies (WWAP, 2017). In low-income countries, only 8% of wastewaters undergo treatment, and globally over 80% of wastewaters are discharged without treatment (WWAP, 2017), contributing to environmental degradation (see Chapter 5). In Europe and North America, as much as 50% of sewage sludge is processed for agricultural use (Nizzetto et al., 2016). Currently, there are 33 megacities in the world, with populations >10 million (United Nations, 2019). With the global trend towards greater urbanisation, P will be increasingly concentrated in urban regions due to food consumption and excretion (Powers et al., 2019). Whilst this carries an increased risk of point sources of pollution, it also represents opportunities to upscale P recycling within coupled agri-urban food systems. However, currently, most cities do not take full advantage of this potential (Metson et al., 2015). For example, in Montreal, Canada, only 6% of the P in municipal waste streams is recycled (noting municipal wastes include more than human excreta wastes, e.g. food wastes, grass cuttings)[i]. There are few studies to evidence the potential for urban P recycling, this is probably the reason why recycling of P in coupled agri-urban food systems does not feature more heavily in city plans. Therefore, if more assessments of this opportunity were carried out, including evaluation of economic and environmental costs compared to the current approach, this might change (Metson and Bennett, 2015).

[i]A full list of the components considered in the definition of municipal wastes is available in the European Commission guidance on municipal waste data collection (European Commission, 2017).

A further complication is that farmers may choose not to use some organic materials as fertilisers because of negative perceptions over the safety of their use in food production. This may be because they fear that consumers will not want to buy their products and/or because they can lose certain certifications for farming practices required by consumers (discussed below) (Bengtsson and Tillman, 2004; Moya et al., 2019a). This concern is especially evident for organic materials derived from human excreta, even if appropriately processed to ensure they are safe to use as fertilisers (Bengtsson and Tillman, 2004; Metson and Bennett, 2015; Moya et al., 2019a).

When farmers do apply recycled sources of P to soils, inefficient practices can result in the P being subsequently lost to the environment (discussed in further detail in Chapter 5). Phosphorus losses from soil occur through soil erosion or leaching processes. Leaching of P is usually limited due to its low solubility but may be higher in soils saturated with P, or with preferential flow pathways if the waste products are not incorporated (Glaesner et al., 2016). Poor manure management can result in significant P losses to the environment in surface runoff (Kleinman et al., 2011; Chapter 4), and long-lasting P losses from soils that have received repeated excess manure applications (Qin and Shober, 2018) (see Chapter 4). For example, organic sources of P such as slurries and manures are often applied in winter in the EU (van Es et al., 2006) and the USA (Williams et al., 2010) which can coincide with heavy rainfall and with frozen fields (in northern latitudes) leading to increased losses through run-off (Komiskey et al., 2011). In the EU, as enforced under the Nitrates Directive of the European Union (91/676/EEC; Council of the European Communities, 1991), areas of land that drain into waters affected by nitrate pollution can be designated as Nitrate Vulnerable Zones (NVZ). Farmers in NVZs are required to comply with measures laid out in action programmes designed to restore water quality, which may include:

- reducing the amount of fertiliser applied;
- prohibiting application of fertiliser during the winter when runoff is greatest and uptake by plants at a minimum; and
- changing the times when animal manures aree applied to the land and holding manures and slurries in tanks until application.

Whilst such legislation focuses on N pollution, it can serve to also reduce P pollution (Amery and Schoumans, 2014).

Challenge 6.2: Manure and waste production is often 'decoupled' from croplands where it can be recycled

In many regions, the distances between the production of phosphorus-rich organic materials and arable land are increasing, driven by the expansion of specialised and intensive farming, urbanisation, and globalised trade. This can make transporting such materials to areas where they can be recycled prohibitively expensive. Decoupling of livestock and arable farming systems is particularly problematic for farmers producing organic foods and feeds. This is because 'conventional' mineral phosphorus fertilisers, and in some cases manures from confined animal feeding operations, cannot be used to fertilise organic crops.

Temporal and spatial separation between sites of P accumulation (livestock farms and cities) and P demand (croplands) means P-rich organic materials must often be stored for long periods and/or transported long distances before use (Metson et al., 2016). Intensive and specialised farming is decoupling arable and livestock systems, and can result in crops being grown increasingly in areas that are not close to livestock (Gerber et al., 2005; Sutton et al., 2013; Lemaire et al., 2014; Watson et al., 2019).

Intensive livestock production systems are expanding rapidly globally, especially in Latin America, and East and South-East Asia (Stenfield et al., 2006; Menzi et al., 2010). The geographical concentration of livestock in areas with little or no arable farming can result in stockpiling of manures (Tamminga, 2003; Menzi et al., 2010), manure mismanagement and P losses leading to pollution of waterbodies (Tamminga, 2003; Gerber et al., 2005; Watson et al., 2019; Glibert, 2020). Consideration of scale when assessing livestock density is important. A region or country may appear to have a low overall livestock density, which conceals areas of high livestock density at smaller scales, and in which animal wastes can quickly exceed the local carrying capacity of the landscape (Tamminga, 2003). For example, the Netherlands farms more cows, chickens, and pigs than any other country in the EU, with 80% of dairy farms producing more animal manure than they can recycle on their land. Strict limits on the application of manure to croplands have been imposed in the Netherlands since 1998, mainly due to ammonia emissions (Erisman et al., 2005), although unpleasant odours can also significantly impact local communities (Sutton et al., 2013), and hence influence policy. Farmers pay an estimated €550 million each year for manure removal, although, it has been reported that farmers are avoiding costs, with up to 40% of manures spread illegally (Dohmen et al., 2017).

Simultaneously, arable farm systems are increasing in size. For example, in the USA there is a trend toward larger farm sizes, with the median USA cropland area on farms almost doubling from 1982 to 2007, from 238 to 447 cropland ha per farm (MacDonald et al., 2013). The availability of mineral P fertilisers, relative to organic P fertilisers, has allowed arable farming to increase dramatically, with low recycling of nutrients from livestock or human waste products (Gerber et al., 2005; Watson et al., 2008, 2019). In some regions, such as Asia and North and Central America, arable farming has become highly reliant on mineral P fertilisers to replenish the P removed in the harvest (MacDonald et al., 2011), and has contributed to a reduction in soil fertility (Watson et al., 2019).

Decoupling of organic livestock and organic arable farming systems is particularly problematic for farmers producing organic foods and feeds. This is because 'conventional' mineral P fertilisers cannot be used to fertilise organic crops in order to meet regulations and fulfil 'organic food' certifications from most international organic food associations (e.g. Demeter, Bioland, Naturland) (Stabenau et al., 2018). In this context, 'conventional' mineral P fertilisers include diammonium phosphate, monoammonium phosphate, single superphosphate, and triple superphosphate. In the EU, this is regulated under European Commission Regulation (EC) No. 889/2008 (European Commission, 2008b) and in the USA, under the National Organic Program (US Government, 2020). In some regions, such as the EU, regulations also recommend the use of organically produced animal manure (i.e. manure from animals fed only organic feedstuffs) but allow the use of conventionally produced manure, provided that it is not the output of 'factory farming' (European Commission, 2008b). Alternatively, in the USA, manures from conventional systems are allowed in organic production, including manure from livestock grown in confinement and from those that have been fed genetically engineered feeds (Coleman, 2012; US Government, 2020).

In a study of 28 organic farms, Foissy et al. (2013) demonstrated that organic farms without livestock, or access to sufficient manures, were depleting soil P and were therefore unsustainable. The alternative for organic farmers is to use fertilisers made with P recovered from wastes[i] (e.g. food wastes, seaweeds, biochar, products or by-products of animal origin[ii]), but these can be expensive and difficult to source. Ground phosphate rock 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).

The global trend of urbanisation (United Nations, 2019) is increasing the distance between centres of human excretion and agriculture (Metson et al., 2015, 2016). Similarly, globalised trade is increasing the disconnect between food and feed production and consumption at international scales (Fader et al., 2013; Hamilton et al., 2018). Some countries are becoming increasingly dependent on food imports due to land and water constraints, with 16% of the world population reliant on international trade to cover their demand for agricultural products (Fader et al., 2013). Multiple countries now consume more crop products than they could produce domestically (even under scenarios of arable land expansion and increased water use efficiency) (Fader et al., 2013). The global flows of P make 'closing the P' loop increasingly complicated as it requires balancing the P imported (e.g. in foods, feeds and fertilisers) with the P exported at the country/regional scale (Withers et al., 2015; Hamilton et al., 2018).

[i]For the EU, a full list of fertilisers, soil conditioners and nutrients permitted for use in organic farming systems is provided in Annex 1 of the European Commission Regulation (EC) No. 889/2008. https://www.legislation.gov.uk/eur/2008/889/annexes

[ii]Under the European Commission Regulation (EC) No. 889/2008, products or by-products of animal origin include blood, bone, and fish meal, and must not be applied to edible parts of the crop.

Challenge 6.3: The reliability of phosphorus-rich organic materials is often lower than mineral fertilisers

The concentrations of phosphorus in organic materials are variable, not easy to determine quickly and lower than mineral phosphorus fertilisers, representing a challenge for farm-scale nutrient management. The bioavailability of phosphorus in organic materials varies and influences their performance as fertilisers, and can be affected by soil type, pH, and crop breed. The bulky nature of many organic materials can make them difficult to spread consistently, affecting their reliability as a fertiliser.

To optimise P applications to soils for maximum plant uptake and minimal losses (i.e. fertiliser 'reliability') farmers need accurate information on the P content of the fertilisers they use and the P content of their soils. However, the concentration of P in some organic materials is highly variable. For example, the P concentration in manures (by dry weight) can range from 4 to 26 g P kg^{-1}, whilst in human excreta, it can range from 5 to 38 g P kg^{-1}. In comparison to mineral P fertilisers, which contain ~200 g P kg^{-1}, most unprocessed P-rich organic materials have relatively low P concentrations (Roy, 2017) (Figure 6.3).

Good agricultural practices adopted across many EU member states aim to limit nutrient applications to optimise crop uptake and include guidance on sludge and manure application to soils (Liu et al., 2018). However, the P content of P-rich organic materials is not easily determined meaning that farmers have low confidence in application rates to meet crop demands. The bulky nature of many organic materials, compared to mineral P fertilisers, can make it difficult to spread, which can also reduce fertiliser reliability (Westerman and Bicudo, 2005).

A further consideration for farmers using P-rich organic materials as recycled fertilisers is the proportion of P that is immediately bioavailable to plants (see Chapter 5). The bioavailability of P is variable between and within the different organic materials but is often not considered in substance flow analyses when assessing opportunities to recycle P (Hamilton et al., 2017). Relative Agronomic Efficiency (RAE) is an estimate of the fraction of P in organic material that will enter the readily available soil P pool and substitute for water-soluble mineral P fertiliser. In assessing RAE, soil P stocks are divided into a readily available P pool (the P immediately available to plants) and a residual P pool (from which P becomes available to plants only through microbial or chemical processes) (Hamilton et al., 2017). Mineral P fertiliser has an RAE of 100% (i.e. all of the P is considered water-soluble and readily available for plant uptake). In contrast, RAE values are lower and variable between P rich organic sources, e.g. cattle manure (82%), sheep/goat manure (75%), pig (77%), poultry (63%), horse (55%), sewage sludge (75%), meat and bone meal (treated with heat and pressure) (19%), food waste (compost) (39%) and food waste (digestate) (55%) (Hamilton et al., 2017).

However, it is important to note that whilst RAE may be a useful indicator, the bioavailability of P in organic materials is heavily impacted by local conditions including soil properties, crop types, drainage, weather, and farming practices such as ploughing (Roy, 2017). The challenge for farmers is, therefore, not only to assess how much P is in the organic material being applied, but also the proportion that is bioavailable for crop uptake, and the rate at which site-specific microbial and chemical processes and soil conditions can convert residual P stocks into bioavailable phosphorus.

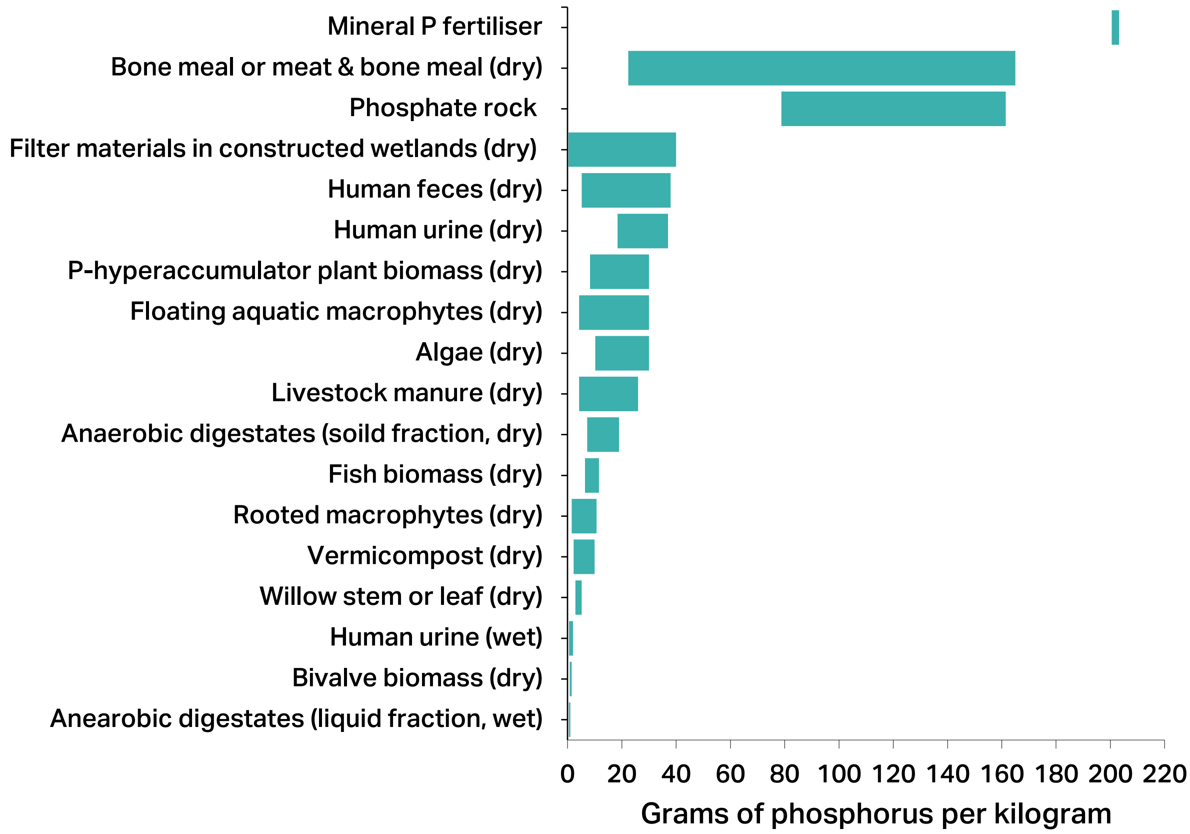

Figure 6.3 Range of phosphorus (P) concentrations (g P kg−1) of different materials, including mineral P fertiliser, phosphate rock, and several organic P materials discussed in this chapter and Chapter 7. Data source: supplementary data in Roy (2017).

Challenge 6.4: Some phosphorus-rich organic materials can contain contaminants

Pathogens, hormones, antibiotics, potentially toxic elements, and microplastics can be present in some phosphorus-rich organic materials. It is important to ensure contaminants are removed, destroyed or concentrations reduced to safe levels in any phosphorus-rich organic materials to be used as fertilisers. In some cases, contaminants can accumulate in soils and may pose a risk to human and animal health and environmental quality.

Phosphorus-rich organic materials can contain contaminants including pathogens, potentially toxic elements, hormones, antibiotics and microplastics (Kinney et al., 2008; Ng et al., 2018; Hill et al., 2019). If P-rich materials used as fertilisers are not sufficiently treated, contaminants can persist and accumulate in soils. In some cases, these contaminants can pose a risk to human health and the environment.

Human and animal faeces can contain significant amounts of pathogenic microorganisms, such as *Escherichia coli* (*E. coli*), *Campylobacter*, Salmonella, *Leptospira*, *Listeria monocytogenes*, *Shigella*, *Cryptosporidium*, *hepatitis A virus*, *rotavirus*, Nipah virus and avian influenza virus. The extent to which pathogenic microorganisms from biosolids and manures applied to agricultural soils can survive and adversely affect human and animal health remains uncertain (Laturnus et al., 2007; Cieslik et al., 2015; Malomo et al., 2018). A study of *E. coli* in manure heaps revealed that the pathogen could survive for up to 47 days, 4 months, and 21 months in bovine, aerated ovine, and nonaerated ovine manure, respectively (Kudva et al., 1998). Whilst risks vary between regions, they should not be underestimated in wastes intended for use as organic fertilisers that are not appropriately treated and used to produce food (Malomo et al., 2018). For example, outbreaks of *E. coli* infection have been associated with water and food, directly and indirectly, contaminated with animal manure (Chapman et al., 1997; Cody et al., 1999; Crump et al., 2002; Sharma and Reynnells, 2018).

Concerns regarding the accumulation of potentially toxic elements (PTEs) in soils receiving applications of P-rich organic material are mixed. For example, Deeks et al. (2013) and Pawlett et al. (2015) observed no significant build-up of PTEs in soils applied with biosolids, whilst Guo et al. (2018a) reported PTE accumulation (especially copper and zinc) in soils under long-term application of pig manure. Bloem et al. (2017) reported that PTE concentrations (including lead, cadmium, mercury and arsenic) were generally lower in livestock manure than biosolids. The PTE content of organic P sources will be largely defined by the PTE consumption of the animals and humans producing them. For example, in 31 intensive farming systems in China, most poultry and livestock feeds contained PTE concentrations above 'National Hygienic Standards for Feeds', and hence the corresponding manures were also high in PTEs (Cang et al., 2004). In the EU, PTE concentrations in sewage are regulated under the EU Sewage Sludge Directive

(86/278/EEC). However, in many regions of the world, such as Africa, regulations are lacking in this respect (Tembo et al., 2017; Fijalkowski et al., 2017). Potentially toxic elements in manure originate mainly from feed additives (e.g. copper and zinc to improve feed utilisation, growth promotion and disease prevention) (Bolan et al., 2004). Mineral P fertilisers, especially those made using sedimentary PR, can also contain cadmium amongst other PTEs (see Chapter 2).

Antibiotics (e.g. tetracyclines, sulfonamides, β-lactams and its metabolites such as sulfonamides and macrolides) can be found in livestock manure (Bloem et al., 2017; Mullen et al., 2019; Menz et al., 2019), biosolids (Boxall, 2018; Magee et al., 2018; Barancheshme and Munir, 2019), wastewaters (Sanseverino et al., 2018; Gudda et al., 2020), and aquaculture wastes (Topp et al., 2018). For decades, multiple varieties of antibiotics have been used together in concentrated animal feeding operations (CAFOs) and aquaculture for prophylactic (prevention), metaphylactic (control), and therapeutic (curative) purposes (Van Boeckel et al., 2015; Zaman et al., 2017; Manyi-Loh et al., 2018). A detailed list of veterinary medicines used in livestock farming, including aquaculture, is provided in Tavazzi et al. (2018). Up to 90% of antibiotics administered to livestock are not metabolised and are excreted without change (Thiele-Bruhn, 2003; Kumar et al., 2005; Sarmah et al., 2006). Some antibiotics in manures will degrade during manure storage, with half-lives in the order of days (e.g. β-lactams and macrolides such as tylosin, Kolz et al., 2005; Boxall and Long, 2005). Others may persist for months to years (e.g. oxytetracycline, tetracycline and amprolium), enabling the transfer of some antibiotics from spread manure to the soil, and aquatic environments through runoff (Hamscher et al., 2002; De Liguoro et al., 2003; Song et al., 2007). In a recent review, the antibiotics present in the highest concentrations in raw and treated manures were enrofloxacin, oxytetracycline and chlortetracycline, with a high risk of release into the environment (Ghirardini et al., 2020). The dispersal of antibiotics in untreated manures is contributing to the growth of antibiotic-resistant bacteria (ARB) in soils and wastewaters (Sanseverino et al., 2018; Gudda et al., 2020). This poses a potential human health risk by increasing human exposure to soil-borne ARB and through ARB contamination entering the human food chain (Kumar et al., 2005; Sarmah et al., 2006; Heuer et al., 2011; Bloem et al., 2017; Barancheshme and Munir, 2019). The increasing prevalence of ARB is a pressing and growing clinical challenge (Zaman et al., 2017; Barancheshme and Munir, 2019).

A range of natural and synthetic hormones are used in livestock production to promote animal growth, and in human populations for therapeutic and contraceptive purposes. The potential exists for hormones consumed by humans or animals to pass into sewage sludge, wastewaters, and animal manures. If these materials are then used as agricultural fertilisers, without first being properly treated, hormones can be transferred to soils and adjacent aquatic environments posing an environmental concern (Lorenzen et al., 2004). Exposure of various organisms to exogenous natural and synthetic hormones, including 17β-estradiol, progesterone, testosterone, zeranol, trenbolone, and melengestrol acerate, has been shown to have endocrine-disrupting effects, which include a variety of developmental and physiological effects (Lange et al., 2002). Estrogenic hormones, such as estrone, 17α-estradiol and 17β-ethynylestradiol, have been detected in swine manures at concentrations ranging from 17 to 4728 ng l^{-1}, 8 to 542 ng l^{-1} and 182 to 357 ng l^{-1}, respectively (Cheng et al., 2018; Singh et al., 2019). Multiple studies have shown livestock manures and poultry litter can be a source of estrogenic hormones to the water environment (Hanselman et al., 2003), for example in rivers in China (Yuan et al., 2014), Denmark (Kjaer et al., 2007) and the UK (Johnson et al., 2006). Several studies have shown that estrogenic compounds can damage human and animal reproduction, immune and nervous systems, inducing deformity of reproductive organs (Witorsch, 2002; Safe, 2004; Caldwell et al., 2008). Intestinal and environmental microbes can transform steroids in excrements, but their activity may not be sufficient for rapid and complete elimination of hormonal activity, especially for synthetic hormones (Lange et al., 2002). However, in a review of multiple risk assessments, Jeong et al. (2013) reported that natural steroid hormones, and synthetic hormone-like substances, have negligible human health impacts when used under recommended veterinary practices (i.e. for therapeutic reasons only). That withstanding, Jeong et al. (2013) also noted that hormones and antibiotics are used illegally, as well as legally, for the growth promotion of livestock animals. Future studies on the environmental concentrations, biodegradability, bioavailability and bioconcentration factors of endogenous and exogenous hormones are necessary to come to a better understanding of their potential impacts on human and wildlife health (Lange et al., 2002; Adeel et al., 2017).

Microplastics (MPs; plastic items with the longest dimension <5 mm) have emerged as a global concern due to their ubiquitous presence in the environment and potential interaction with biota (Ng et al., 2018; Wang et al., 2019; Qi et al., 2020; Crossman et al., 2020). Synthetic fibres originating from domestic washing machines are a major source of MPs in sewage (Ziajahromi et al., 2017; Henry et al., 2019). Whilst media attention has identified plastic microbeads in personal care products (included in some kinds of toothpaste, soaps and facial scrubs) as a key source of MPs in wastewaters, Duis and Coors (2016) argue their contribution to the aquatic environment is minimal in comparison to other sources. Fortunately, wastewater treatment plants can efficiently remove over 90% of MPs from wastewaters (Carr et al., 2016; Corradini et al., 2019). However, this concentrates MPs into sewage sludge (Corradini et al., 2019), and does not address the problem in regions where wastewaters are discharged without treatment (see Chapter 5). As discussed earlier, the use of sewage sludge as fertiliser for agricultural applications is often economically advantageous and is common in many developed regions (Nizzetto et al., 2016). In some regions, the application of biosolids to soils may represent a significant source of MPs to agricultural systems since biosolids can contain up to 1.4 x 104 MP particles kg^{-1} (Crossman et al., 2020). In the EU, the USA, China, Canada and Australia, approximately 26,000, 21,000, 14,000, 1,500 and 1,000 t $year^{-1}$ of MPs, respectively, are added to farmlands in biosolids (Mohajerani and Karabatak, 2020). Data on the existence and transfer processes of MPs in soils are much less available than for aquatic environments, and the implications of MPs for the soil environment and consequences for food security require further assessment (Nizzetto et al., 2016; Wang et al., 2019). While agricultural soils may be among the largest environmental reservoir for MPs, studies assessing the scale of contamination are conspicuously absent (Nizzetto et al., 2016). However, concerns have been raised that MPs may act as vectors for other forms of pollutants, such as PTEs and endocrine disruptors, aiding their accumulation in soils (Turner and Holmes, 2015; Nizzetto et al., 2016; Wang et al., 2019; Qi et al., 2020). Furthermore, MPs can break down and create nanoplastics (NPs), which, due to their smaller size (<1 μm), can be absorbed by plant cells. Nanoplastics can decrease microbial mass and enzyme activity within soils (Mohajerani and Karabatak, 2020). Preliminary lab studies by Bosker et al. (2019) showed nanoplastics can accumulate in seed capsules and provide a short-term and transient delay in germination and root growth. Whilst data for impacts of MPs in soils is growing, the transfer of MPs from land to aquatic systems is already acknowledged (Ng et al., 2018; Wang et al., 2019; Qi et al., 2020; Crossman et al., 2020). In a study of three agricultural fields in Ontario, Canada, receiving applications of biosolids containing MPs, >99% of MPs applied from biosolids were estimated to be exported to the aquatic environment (Crossman et al., 2020).

The issues above highlight a key challenge: how to ensure that the recycling of manure and biosolids, as a key P sustainability measure, does not contribute to an increase in human exposure to PTEs, pathogens, ARBs, antibiotics, endocrine disrupters, and MPs and their dispersal and accumulation into the environment.

Challenge 6.5: Policy, infrastructure, and financial support are lacking for phosphorus recycling

There is a lack of coordinated policy and regulation to support an increase in the recycling of phosphorus-rich organic materials. In some regions, there is little economic incentive for farmers to switch from mineral phosphorus fertiliser to phosphorus-rich organic materials. Some farmers can face legal and certification barriers stopping them from recycling certain phosphorus-rich organic materials.

Farmers often work within narrow profit margins and may not be willing to take the risk of reducing their mineral P fertiliser application rates in favour of P-rich organic materials (or recovered P fertilisers), unless it is guaranteed to be equally or more profitable (e.g. supported by subsidies) or enforced through policy and regulation (Kleinman et al., 2015). In comparison to the use of P-rich organic materials as fertilisers, the tools, knowledge, and infrastructure to support mineral P fertiliser use are well established (Sommer et al., 2013; Case et al., 2017). As a result, in many regions, it can be cheaper and/or easier to fertilise soils with mineral P fertiliser than most P-rich organic materials. That withstanding, for farms that integrate crops and livestock, animal manures are available at no cost beyond the cost of land application, whilst mineral P fertilisers must be purchased. However, managing urine and liquid manure, especially in non-mechanised situations and on smaller farms, requires investments in infrastructure and innovation and can be further constrained by labour requirements (Teenstra et al., 2014). In low-income countries, a lack of access to credit for simple manure storage and application equipment remains a key barrier for P recycling (Teenstra et al., 2014). Investment is needed to provide producers of P-rich wastes (e.g. livestock farmers, abattoirs) better access to organic waste processing facilities, such as building local facilities or improving waste collection and transport systems. As acknowledged above, investment is also needed to improve transport infrastructure where it may be limiting farmer access to P-rich organic materials (Case et al., 2017).

Manure is the most prevalent source of P-rich organic material so efficient P recycling is underpinned by sustainable manure management. Although manure management policies are common in Asia, Africa and Latin America, enforcement has been weak (Teenstra et al., 2014). This was particularly apparent where multiple regulatory bodies were involved, resulting in a complex regulatory system and, as a result, policy incoherence. Nevertheless, the absence of a mandatory manure policy does not indicate the absence of good manure management practices (Teenstra et al., 2014). For example, despite the absence of manure policies in El Salvador, farmers routinely apply manures to crops and some large farms use biodigesters to process manure and apply digestates to soils. Indeed, schemes for better nutrient use efficiency on farms are often voluntary, as seen in many farming communities in the USA, where a combination of volunteer and litigated nutrient management strategies have been applied (Sharpley et al., 2012; Kleinman et al., 2015). However, whether imposed by regulation or adopted voluntarily, the success of these strategies relies on having adequate local information and stakeholder support (Kleinman et al., 2015). A lack of appropriate information can result in policies that do not reflect the needs of the local communities. This was observed in a policy in Arkansas USA, where mandated manure export from poultry farms (for environmental reasons), adversely affected beef producers who had to purchase extra mineral P fertiliser to meet the shortfall in poultry litter they had previously used to fertiliser their pastures (Kleinman et al., 2015).

In some cases, some farmers can face legal and certification barriers stopping them from recycling some P-rich organic materials, for example, the regulations regarding the use of human wastes in the production of organic foods for human consumption. Some countries do not allow any use of human wastes (e.g. the USA, EU countries, Uganda), others prohibit the use of sewage sludge but allow the use of human excrements on non-edible crops (e.g. Mexico), while other countries prohibit the use of untreated human excrements but allow the use of treated sewage sludge (e.g. India, Australia) (Seufert et al., 2017). Similarly, the most widely adopted standard for quality assurance of horticultural crops, GLOBALG.A.P[i], withholds certification for Good Agricultural Practices (G.A.P.) if farmers producing fruits and vegetables apply biosolids to their soils, even if safety protocols are followed to reduce human and animal health risk (Moya et al., 2019a).

[i]GLOBALG.A.P. (formally EurepGAP) is a farm assurance programme that audits farms and agricultural products, with an internationally recognised set of farm standards dedicated to Good Agricultural Practices (G.A.P).

6.4 Solutions

Solution 6.1: Treat waste streams as valuable nutrient resources

A paradigm shift in how we view phosphorus-rich waste streams is needed; from pollutant to valued nutrient resource. Key actions in delivering this shift include raising awareness of the costs of phosphorus losses and benefits of phosphorus recycling, providing education and extension services to encouragestakeholders to recycle phosphorus, and mobilising investment in infrastructure and technology to make phosphorus recycling safe, easy, and efficient.

A paradigm shift in how we regard our waste streams is required; waste products should not be wasted products. Quantifying the economic benefits of recycling P-rich organic materials is critically important to support the decisions of governments, stakeholders, and the public. Such estimates should consider the costs of mineral P fertilisers that can be replaced by recycled P sources, as well as the agronomical and environmental benefits of recycling P-rich organic materials. However, these should be offset against the costs of processing, storing, and transporting P-rich organic materials. For example, in 2013, the value of P lost globally in animal manures (~4 Mt P) and human excreta and other human wastes (~6 Mt) (as estimated by Chen and Graedel, (2016)), expressed as cost per unit P in fertiliser, is estimated at US$12.8 and US$19.2 billion, respectively. This is based on a cost of P in diammonium phosphate (DAP) of US$3.2 P kg^{-1} (for September 2021), and assumes all losses are replaced by DAP.[i]

Global advocacy, ambition, dialogue and awareness-raising of the environmental benefits of P recovery and recycling will help to improve public and political support (Matsubae and Webeck, 2019). The perception that fertilising agricultural soils with mineral P fertilisers is safer and more reliable than recycled P products persists in farmer communities (Case et al., 2017). In an extensive analysis, Piñeiro et al. (2020) observed that independent of incentive type, one of the strongest motivations for a farmer to adopt a sustainable behaviour is the perceived benefit for their farm or the environment. Similarly, it has been demonstrated that behavioural factors should be considered in economic analyses of farmer decision-making and are important in developing more realistic and effective agri-environmental policies (Dessart et al., 2019). A component of changing some of these habitual behaviours is the provision of knowledge and tools to support decision making on the collection, storage, processing and recycling of P-rich organic materials at the farm scale. This may be delivered by extension services, government agencies or through peer-to-peer knowledge exchange (Brownlie et al., 2015; Drangert et al., 2017; Aregay et al., 2018). Farmer to farmer communication and farmer champions that can advocate the benefits of recovered P fertiliser will be important in raising

[i]Data from https://blogs.worldbank.org/opendata/fertilizer-prices-expected-stay-high-over-remainder-2021. It is assumed DAP contains 46% P_2O_5; therefore, DAP has a ~20% P content. With substantial fluctuations in DAP price (e.g. ranging from US$280-643 DAP t^{-1} between 2010 to 2021) this value varies greatly.

awareness (Brownlie et al., 2015; Backus, 2017). Flagship farms in the Netherlands have helped to demonstrate to the farming community how environmental measures can be implemented in real operating conditions (Backus, 2017). In the developed world, sub-optimal recycling of P-rich organic materials is often linked to the level of education of many small-scale farmers, and the lack of infrastructure to support farmers to access available knowledge (e.g. literacy remains an issue in some regions) (Teenstra et al., 2014). Teenstra et al. (2014) highlight that knowledge development is not a one-off intervention and will require continuous programmes, with a significant shift in educational approaches and the development of frameworks that provide long-term interdisciplinary support (Reitzel et al., 2019).

Increased utilisation of low-tech methods to recycle P from human wastes is needed, especially in low-income countries, where access to mineral P fertilisers and funding to support is limited. Whilst in many low-income countries the most critical driver for improving sanitation is health risks, maximising the opportunities for the safe recycling of P in human excreta and wastewater is a win-win (Trimmer et al., 2017). Pilot projects that collect human wastes and process them into fertilisers have been implemented in Kenya, Madagascar, South Africa and Ghana (Cofie et al., 2009). These schemes address the sanitation challenge, reduce water pollution, develop business growth and provide a cheap source of P fertiliser to farmers (Cofie et al., 2009). Whilst the technologies are known, mobilising investment in infrastructure and equipment to set up such projects often remains the greater challenge. In regions where sewerage is limited, simple methods to separate the collection of faeces and urine (e.g. urine-diverting toilets - Figure 6.4) may offer an opportunity to recycle the P in urine (Yadav et al., 2010; Mihelcic et al., 2011; Moya et al., 2019b). Whilst faecal matter should be treated to remove pathogens, urine can be applied directly to soils, safely, as a fertiliser. It is estimated that the urine produced by a single person in a year contains enough P to fertilise a crop area of 400 m^2 for a growing season (Mihelcic et al., 2011). Assessments in both southern India (Simha et al., 2018) and South Africa (Wilde et al., 2019) showed that consumers had mostly positive attitudes towards using recycled human urine as a fertiliser. The use of black soldier fly larvae shows great promise as a sustainable and low-tech method to process solid P-rich organic materials (e.g. food wastes, livestock residues and biosolids) into animal feed and fertilisers (Dicke, 2018; Shumo et al., 2019) (Figure 6.5).

Policies should focus on limiting the losses of potentially valuable organic P residue streams to landfills and the inclusion of P-containing ashes in building materials such as cement. These are outdated methods to deal with resource-rich residue streams, highlighted by the relatively recent emergence of landfill mining as a means for procuring renewable raw materials (Schreck and Wagner, 2017).

Figure 6.4 Urine, free from feacal contamination, can be used as a safe source of phosphorus fertiliser for crops. **a)** Urine diverting toilets in Nepal. Urine is collected in the basin at the front, whilst feaces collect in the hole at the back. **b)** Left: urine-diversion flush toilet by Roediger (Germany). A valve opens only when the user is seated to prevent flushing water from draining into the urine tank. Right: urine diversion flush toilet by Gustavsberg (Sweden). In this design no valve is used, this allows a little bit of flushing water to enter the urine pipe to avoid potential clogging of assemblies. Photographs courtesy of The Sustainable Sanitation Alliance (SuSanA).

Figure 6.5 Rearing black soldier fly larvae in Cameroon. Black soldier flies/larvae can be fed on a range of municipal wastes facillitating the recovery of phosphorus through biological assimilation. The larvae can be fed to livestock allowing the safe recycling of phosphorus and providing an inexpensive and sustainable livestock feed. Photograph courtesy of the International Institute of Tropical Agriculture (IITA).

Solution 6.2: Optimise the spatial integration of arable and livestock agricultural systems

Landscape planning can integrate arable and livestock farming to maximise nutrient recycling. Whilst efforts should be made to ensure animal densities in livestock farming do not exceed nutrient needs, some farming systems must rely on disposal/utilisation contracts. Arable-livestock farming partnerships can support the exchange of crops, grains, and manures, and coordinate land-use to support more regionally closed feed-manure loops.

Landscape planning should consider the integration of arable and livestock farming to maximise nutrient recycling (Sutton et al., 2013). In practice, this means livestock farmers should be able to recycle their manures and livestock residues efficiently. Ideally, to ensure a sustainable animal production system, animal densities should be selected to ensure that the nutrient requirements of the local crops are not exceeded (Tamminga, 2003; Erisman et al., 2011). A crucial question then becomes what is the P carrying capacity of the soil and, if exceeded, what options are available to recycle manures elsewhere, e.g. via manure disposal/utilisation contracts.

Concentrated animal feeding operations (CAFOs), like those of pigs and poultry, have no other option but to rely upon manure disposal contracts, or export contracts, usually with arable farmers, which confirm manures will be exported from a farm and imported onto another farm or to another operation (e.g. anaerobic digester) to be processed/utilised (Tamminga, 2003; van Grinsven et al., 2005; DAERA, 2021). Where manures need to be transported long distances, dewatering can significantly reduce volume, and therefore the energy and cost of transportation, whilst also increasing P and N concentration.

For some non-mixed farming systems, that are either only arable or only livestock, arable-livestock farming partnerships may be necessary to support greater recycling (Asai and Langer, 2014; Martin et al., 2016; Moraine et al., 2017; Asai et al., 2018). Local farmer cooperatives can negotiate the exchange of crops, grains and manure, as well as coordinate land-use allocation patterns, collectively planning the crops and animal movements in each field to optimise rotational manure application and crop rotations (Martin et al., 2016; Asai et al., 2018). Whilst this can involve extensive and long-lived coordination between farmers, this in itself can deliver many co-benefits including social benefits and collective empowerment of farmers (Martin et al., 2016). Such community-based schemes are best supported by collective participatory workshops involving farmers, agricultural consultants and researchers (Martin et al., 2016). For some organic farmers, the development of collaborative partnerships to exchange organically produced feed and manure is essential and contributes to adaptability and flexibility against a backdrop of tightening regulations (Asai and Langer, 2014). Balanced nutrient budgets are commonly observed in organic farming systems producing large quantities of manure or which purchase organically approved feed (Wivstad et al., 2005; Foissy et al., 2013). That withstanding, in a study of 23 organic farms in southern France, P budgets for farms that did not have livestock or import manures were not necessarily negative, and opportunities existed to further optimise nutrient cycling within the farm (Lamine and Bellon, 2009; Nesme et al., 2012; Foissy et al., 2013).

In certain areas or to address certain problems, a reduction in the number of animals seems inevitable (Tamminga, 2003). In such cases, farmers may need support to diversify their agricultural outputs. For example, where livestock farming is the dominant output in a given catchment and where this activity results in water quality impairment (e.g. the Chesapeake Bay and the Lake Okeechobee catchments in North America, the Po Delta in Italy and the sand regions of the Netherlands (Greaves et al., 2010)), legislation may be required to control P recycling to land (Erisman et al., 2011). However, the magnitude of livestock reduction may be influenced by optimising nutrient management elsewhere in the farming system, displacing the P produced in one catchment to another to balance across scales (Tamminga, 2003). In simple terms, nutrient inputs should equal exports. This can be achieved by reducing inputs, increasing exports, or a combination of both, and requires spatial planning with respect to the carrying capacity of the system (Greaves et al., 2010).

Integration of cropping and livestock systems also supports habitat diversity and increases the adaptability of farming systems to cope with socio-economic and climate change-induced shocks (Lemaire et al., 2014). However, to implement the measures suggested above will require legislation, but also acceptance by the farmers as important stakeholders. For the latter, education and financial incentives will be required (Tamminga, 2003). Furthermore, relocating and dispersing livestock systems has major implications for supporting infrastructure and delivery mechanisms for products to consumers.

Where dense human populations, animal populations, and croplands occur adjacently, many large P flows converge within a relatively small locus. These areas disproportionately influence the contemporary global P system and thus are hotspots for P recycling (Powers et al., 2019). Developing a better understanding of large-scale nutrient flows and related policies will help to identify and better manage spatially disproportionate nutrient losses and impacts (Bergström et al., 2008; Hamilton et al., 2018).

Solution 6.3: Utilise available technology and tools and provide education

The reliability of phosphorus-rich organic materials as fertilisers can be improved by processing to improve fertiliser quality, and developing better systems to help farmers assess the phosphorus content and phosphorus bioavailability of the materials. Furthermore, farmers can be better supported to optimise the application of recycled phosphorus products and other nutrients in order to maximise phosphorus uptake by plants. However, critical to this is a sufficient understanding of farm- and local-scale nutrient budgets.

Processing methods that can increase the fertilising qualities of P-rich organic materials should be further developed and better utilised (e.g. practices that increase P concentration, P bioavailability and reduce contaminants). However, processing costs should be recoverable otherwise there is a significant disincentive to farmers to adopt them. Low technology processes include dewatering of manure to produce a solid fraction with relatively high P content and low water content, reducing volume and hence transport costs (Møller et al., 2000). This can be carried out in large-scale central installations, or small-scale mobile installations. Other low technology processes used to improve the fertiliser quality of multiple P-rich organic materials are summarised in Table 6.1.

Farmers applying P-rich organic materials to their soils may find it difficult to achieve reliable fertiliser application rates if the P content and bioavailability of the material is highly variable (Figure 6.2). The development of an assessment and reporting system, that can be used by farmers, to share detailed information on the P content and P bioavailability of organic materials, may support more accurate management of P inputs. This could be developed through a relevant international body (e.g. FAO or UNEP), with public access provided, for example through a web-based data portal/database. Using manures as an example, such a tool should detail how the P content in manures varies between animal species, of different ages, for a range of dietary P inputs, with and without phytase dietary additives, and from different farming systems (e.g. meat or dairy production). Further important information for farmers would be how different processing and storage methods and soil types can potentially impact P bioavailability of P-rich organic materials, as well as losses of other nutrients, such as N losses through ammonia volatilisation or nitrate leaching (Nicholson et al., 2002; UNECE, 2014). Compiling these data (much of which already exists in the literature) into an online user-friendly database would be a useful tool, for example, to help farmers calculate the right amount of manure to apply at the right time to match plant nutrient demands for P and other macro and micronutrients.

Free availability of such information is especially important, as tools to make in-field assessments of the P content and P bioavailability of organic materials are not commonly used and may be limited to higher technology farming systems (Lugo-Ospina et al., 2005). The development of dry spectral techniques such as portable X-ray fluorescence analysis and mid-infrared spectroscopy may prove more useful by providing more accurate information on the P content in organic materials (Vogel et al., 2016; López-Núñez et al., 2019), but are also not widely used, due to the expense (i.e. equipment is >US$7000) and the specialist expertise required for operation and interpretation of the results. However, using such tools to provide comparable data in the field can be challenging. Laboratory analyses are likely required to provide meaningful data for farmers and should be produced using batch testing and standard and accredited analytical approaches.

Alongside this information, farmers need to be able to accurately assess their farm- and local-scale nutrient budgets, to identify opportunities to increase their use of recycled P fertilisers. This requires the collection of accurate data on P inputs, outputs, and stocks using agreed data collection criteria/system boundaries (Rose et al., 2016). Farmer advisory services can support farmers to record accurate data on P inputs (e.g. feed and fertiliser use) and exports (e.g. harvests). A range of soil sampling methods can be used to assess P stocks in soils (Knowles and Dawson, 2018; Lawrence et al., 2020). Several software decision support tools (DSTs) are available that can analyse such data and generate evidence-based recommendations (for an extensive list see Drohan et al., 2019). Farmer advisory services remain critical in facilitating the use of relevant DSTs within farming communities. However, uptake of such tools is often low due to factors that include poor usability, cost-effectiveness, performance, and relevance to the user and their local conditions (Rose et al., 2016). Technological advancements in monitoring, satellite imaging, sensors, remote sensing, and analytical instrumentation will facilitate the development of DSTs that can incorporate extremely large data sets (i.e. 'big data'). Such DSTs may be increasingly able to identify heterogeneity in local conditions, over wider geographical areas, making them more useful for the farmer (Drohan et al., 2019). Several 'user-friendly' DSTs are already available to help farmers make more effective decisions on nutrient application rates. For example, 'the Farm Crap App' is a manure management app (https://www.swarmhub.co.uk/the-farm-crap-app-pro/) that provides an easy to use, and accurate and reliable way to manage and record slurry spreading information and data on manure. Another example is the 'Phosphate Acceptance Map' (PAM), a novel tool for assessing land suitability for biosolids application at a national scale (Wadsworth et al., 2018).

In the food system, the farmer is at the forefront of daily decision making in P management (Drohan et al., 2019). If the P content and P availability in organic sources can be accurately determined, then this can be used alongside other critical information (e.g. soil types and P content, crop type, field slope) to determine the optimal amount of organic material to apply to agricultural soils to maximise plant growth and minimise P losses.

Solution 6.4: Process organic materials appropriately and provide safety certification schemes

Most phosphorus-rich organic materials need some processing to reduce contaminants and pathogens to safe levels for use in food production. Reducing livestock dietary intake of potentially toxic elements and imposing strict limits on the non-therapeutic use of antibiotics in livestock, will reduce levels of these contaminants in manure and biosolids. Assurance that fertiliser products derived from phosphorus-rich organic materials are safe for their intended use should be provided to end-users.

Dietary intake of contaminants by livestock should be reduced to decrease levels in manures and biosolids (Cang et al., 2004). Furthermore, imposing strict limits on the non-therapeutic use of antibiotics in livestock should be implemented as a global priority (Barancheshme and Munir, 2019). In 2017, the U.S. Food and Drug Administration banned the use of antibiotics in livestock without a prescription from a veterinarian and made it illegal to use drugs solely for growth promotion (FDA Center for Veterinary Medicine, 2018). Similarly, in the EU, from 2022, new legislation (Regulation (EU) 2019/61 on Veterinary Medicines and Regulation (EU) 2019/4 on Medicated Feed) will prohibit all forms of routine antibiotic use in farming, including preventative group treatments. However, antibiotics are still routinely added to livestock feeds in many parts of the world (Van Boeckel et al., 2015). In 2015, 97,000 t of antibiotics were used in animal husbandry in China (Collignon and Voss, 2015). Globally, antimicrobial drug consumption is projected to rise by 67% by 2030 (from 2010 levels), and nearly double in Brazil, Russia, India, China, and South Africa (Van Boeckel et al., 2015). Until antibiotic use is better regulated in these countries, steps should be taken to ensure manures are appropriately treated to ensure their application to cropland does not contribute to the proliferation and human exposure to ARB.

Contaminants in human biosolids and wastewaters can be further reduced through changes to domestic and industrial behaviours. For example, reduction in plastic use followed by the mainstream adoption of 'compostable' and 'biodegradable' films may help reduce MPs contamination in some P-rich organic materials and reduce soil MPs accumulation (Song et al., 2009; Qi et al., 2020). Encouragingly, policies to restrict single-use plastics have been announced in many parts of the world, including in Canada, the USA, and several countries in the EU and Africa (Xanthos and Walker, 2017). Guerranti et al. (2019) call for a global ban on the use of polymeric microbeads in cosmetics and personal care products. Such regulations already exist in some countries (e.g. the EU and the USA), and may indirectly reduce MPs entering wastewater treatment plants and thus reduce concentrations in sludge and biosolids (Crossman et al., 2020). However, policies are needed to address environmental pollution from MPs originating from the laundry of synthetic clothes, which will be unaffected by such legislation (Crossman et al., 2020). Some contaminants, such as medical pharmaceuticals, may not be possible to eliminate, but can instead be degraded in downstream processes.

Many pathogens and antibiotics in organic materials can be destroyed by composting, and even more by vermicomposting (Dolliver et al., 2008; Wang et al., 2016; Soobhany et al., 2017). For example, holding swine and cattle manure at 25°C for 3 months will render it free from key, common pathogens, including *E. coli*, Salmonella, *Campylobacter, Yersinia, Cryptosporidium*, and *Giardia* (Guan et al., 2003). Zheng et al. (2008) reported that low tech processing of wastes, such as multi-stage lagoon systems, and increasing manure-piling times, can promote degradation processes of pharmaceuticals and hormones (in particular). Although thermo-chemical processes such as anaerobic digestion also reduce pathogen levels, some antibiotics and ARB can persist under such conditions and high/thermophilic temperatures may be required to significantly degrade them (Youngquist et al., 2016). However, the most reliable method to destroy all organic contaminants is incineration or thermal gasification; P can then be recovered from the end products (e.g. ashes). Processes to recover P from waste streams are discussed in greater detail in Chapter 7.

Assurance that fertiliser products derived from P-rich organic materials are safe for use should be provided to end-users, alongside appropriate information on the method used to remove contaminants and pathogens. For example, the Biosolids Assurance Scheme (www.assuredbiosolids.co.uk), which is supported by UK water utilities, provides food chain and consumer reassurance that certified biosolids can be safely and sustainably recycled to agricultural land. Other examples of assurance schemes include ReVAQ in Sweden, the National Biosolids Partnership in the USA and the Australasian Biosolids Partnership (ABP) in Australia and New Zealand, all of which provide a certification scheme for biosolids to increase customer confidence in their use in agriculture (Gale, 2007; National Biosolids Partnership, 2011; l'Ons et al., 2012). Importantly, auditing to ensure biosolids conform to standards should be carried out by an independent third-party certification body.

Solution 6.5: Develop policies, regulations, and financial instruments that support phosphorus recycling

Improved coordination between relevant government bodies and relevant stakeholders is required to develop coherent, holistic policies and create markets for recovered phosphorus fertiliser. Investment in infrastructure and technologies supported by cross-sectorial innovation, co-creation and sharing of knowledge can help to make phosphorus recycling simple and efficient. The economic benefits for society of recycling phosphorus need to be better quantified and used to encourage stakeholders to recycle phosphorus more efficiently. The value of recovering phosphorus can be maximised by selecting methods to process organic materials that produce additional co-benefits.

In most regions, a 'carrot and stick approach' which combines mandatory requirements for change with incentives, will be needed if significant increases in the recycling of P-rich organic materials are to be achieved (Johansson and Kaplan, 2004; Zahariev et al., 2014; Backus, 2017). The 'sticks' in this approach include the development of regulations and policies that set mandatory targets for reducing P losses through sustainable P recycling, as well as the better enforcement of existing related policies (e.g. those for sustainable manure management; Teenstra et al., 2014). Careful monitoring of P flows in agricultural systems are critical to inform policy development, and should be supported by all farmers and relevant stakeholders, through accurate recording of relevant data, e.g. livestock numbers, nutrient content and volume of feed consumed, production systems used, and the use of animal manure and fertilisers.

The 'carrots' in this approach may include subsidies and tax incentives to encourage stakeholders to recycle P from waste streams. In some regions, this should be extended to financial credit to cover capital costs for recycling equipment (e.g. manure spreading equipment, organic waste processing facilities) (Gerber et al., 2005; Mayer et al., 2016). It is likely that in some regions, especially low-income countries, significant investment in infrastructure and technologies to make P recycling simple and efficient (e.g. communal manure storage facilities, better systems and access roads to transport P-rich organic materials to croplands) will also be required. Additional funding and support for research in P recovery and recycling would also further develop the viability of these technologies, in partnership with the private sector (see Chapter 7). Whilst the type and level of support needed will vary between sector and region, stakeholders should be supported to implement the changes needed, without significant hardship, and ideally with economic/production gains. However, a key challenge is identifying funds and the appropriate financer, e.g. farmers, waste treatment industries, food processing industries, and governments (whose funds are ultimately paid by society through taxes). We argue that recycling P-rich organic wastes (as opposed to disposal without nutrient recycling) can provide economic value for all stakeholders, which can be used to support changes directed through policy and regulation.

For farmers, the direct economic value of using P-rich organic wastes to fertilise crops (which are most commonly provided for free), is through greater crop yields, and reduction in expenditure on mineral P fertiliser costs (Withers et al., 2015; Mayer et al., 2016; Leip et al., 2019). Economic value can be maximised by selecting methods to process organic materials that produce additional co-benefits, such as biogas production that can produce renewable energy whilst also recovering N, minerals and metals (Mayer et al., 2016). Currently, biodigester programmes have been launched in more than 50 countries across Africa, Asia and South America, as a way of advancing agricultural productivity, renewable energy use and waste management (Buysman and Mol, 2013; Teenstra et al., 2014; van Hessen, 2014). There are also long-term benefits for farmers, which can be more difficult to monetise, but include resilience to fluctuation in mineral fertiliser costs, and agronomic benefits and improvement to soil health (Mayer et al., 2016). In an assessment of strategies to reduce P losses from dairy farmers in New Zealand and Australia, McDowell and Nash (2012) provide a range of fully costed strategies, allowing farmers (or farm advisors) to choose those that suit the farm system whilst maintaining profitability. A similar approach is needed to help farmers make decisions regarding their options to recycle P-rich organic material. However, local assessments should be made to identify the cost-effectiveness of different strategies and to avoid negative feedback and interactions among management strategies (Smith et al., 2015; Jarvie et al., 2017; Macrae et al., 2021).

For wastewater treatment, food processing and food retail industries, disposal of wastes (without P recycling) can incur significant costs (Kosseva, 2009; Peccia and Westerhoff, 2015; Bai et al., 2016b; CBS, 2020). For example, in the USA alone, disposal of food wastes to landfill is estimated to cost US\$1.0 billion year^{-1} (Kosseva, 2009), whilst US sewage sludge managers report the landfill (tipping fees and hauling), and incineration (hauling, incineration costs) of sewage sludge can cost US\$100–650 dry t^{-1}, and US\$300–500 dry t^{-1}, respectively (Peccia and Westerhoff, 2015). By increasing the conversion of P-rich organic wastes into value-added products (e.g. fertilisers), waste-producing industries may avoid/offset waste disposal costs. The savings made could then be used to support farmers that recycle the P-rich organic materials (Peccia and Westerhoff, 2015; Withers et al., 2018; Tonini et al., 2019). Equally, where the recycling of P-rich organic wastes is accompanied by corresponding reductions in mineral P fertiliser use, the co-benefits should be quantified and used to justify any financial investments that may be needed to optimise P recycling capacity. For example, short-term benefits may include a potential reduction of P losses to waterbodies (assuming P-rich organic materials are sustainably managed) (see Chapter 5). They will also include the reduction in externalities associated with mining PR and the manufacture of mineral P fertilisers, such as emissions to air of fine rock dust and sulfur dioxide and release of potentially toxic elements into the environment (World

Bank, 2007; EPA, 2010; Tonini et al., 2019). A long-term benefit is the preservation of finite PR reserves for the protection of food security for future generations (see Chapter 2). In linear and nutrient 'leaky' systems, such externalities are largely paid for by society, rather than the polluters or nutrient users (Tonini et al., 2019).

With such wide-reaching benefits, it may be applicable for some farmers to be financially supported to recycle P (e.g. by applying P-rich organic materials, or recovered P products/fertilisers to their soils) with corresponding reductions in the use of mineral P fertiliser. The idea of `payments for ecosystem services' (PES), whereby land users (e.g. farmers, forest owners or managers) are paid for the service they provide to the environment is not new. Examples date back to the 1930s when the US government paid farmers to preserve certain types of landscape (Duboua-Lorsch, 2020). In the EU, the future Common Agricultural Policy (CAP) (to be implemented from 2022) will introduce 'eco-schemes' which, like PES, will aim to remunerate farmers engaged in sustainable practices, including certain sustainable P measures (Lampkin et al., 2020). The effectiveness of PES and the EU 'eco-schemes' in achieving environmental objectives (e.g. for P recycling), can be improved by providing strong and quantified objectives, ensuring strategies receiving financial support are appropriate to the local circumstances, and guaranteeing payments are provided for long enough to allow farmers planning security (e.g. if farmers are going to build new infrastructure/invest in processing technologies, then >5-10 years funding may be appropriate) (Börner et al., 2017; Heyl et al., 2020).

Policies, regulations and standards for good agricultural practices should reflect accurate knowledge on the risks of using biosolids as fertiliser, and not provide a barrier if there is no risk to human, animal, or environmental health. Moya et al. (2019) highlight the power of the GLOBAL-G.A.P. quality assurance standards for farmers in low-income countries, and how failure to achieve them by using fertiliser derived from human wastes can affect their ability to export goods to high-income nations. Whilst GLOBALG.A.P. quality assurance standards are not a globally agreed policy, they are highly influential in directing P recycling behaviours and impact those least financially able to replenish their soils with mineral P fertiliser (e.g. small-holder farmers in low-income nations).

Sufficient knowledge and technology are already available to make significant increases in recycling P-rich materials globally. The challenge moving forward is finding the investment, resources and policy support to make the transition to a circular economy for phosphorus (Oenema, 2004; Teenstra et al., 2014; Cordell and White, 2014; Li et al., 2015).

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