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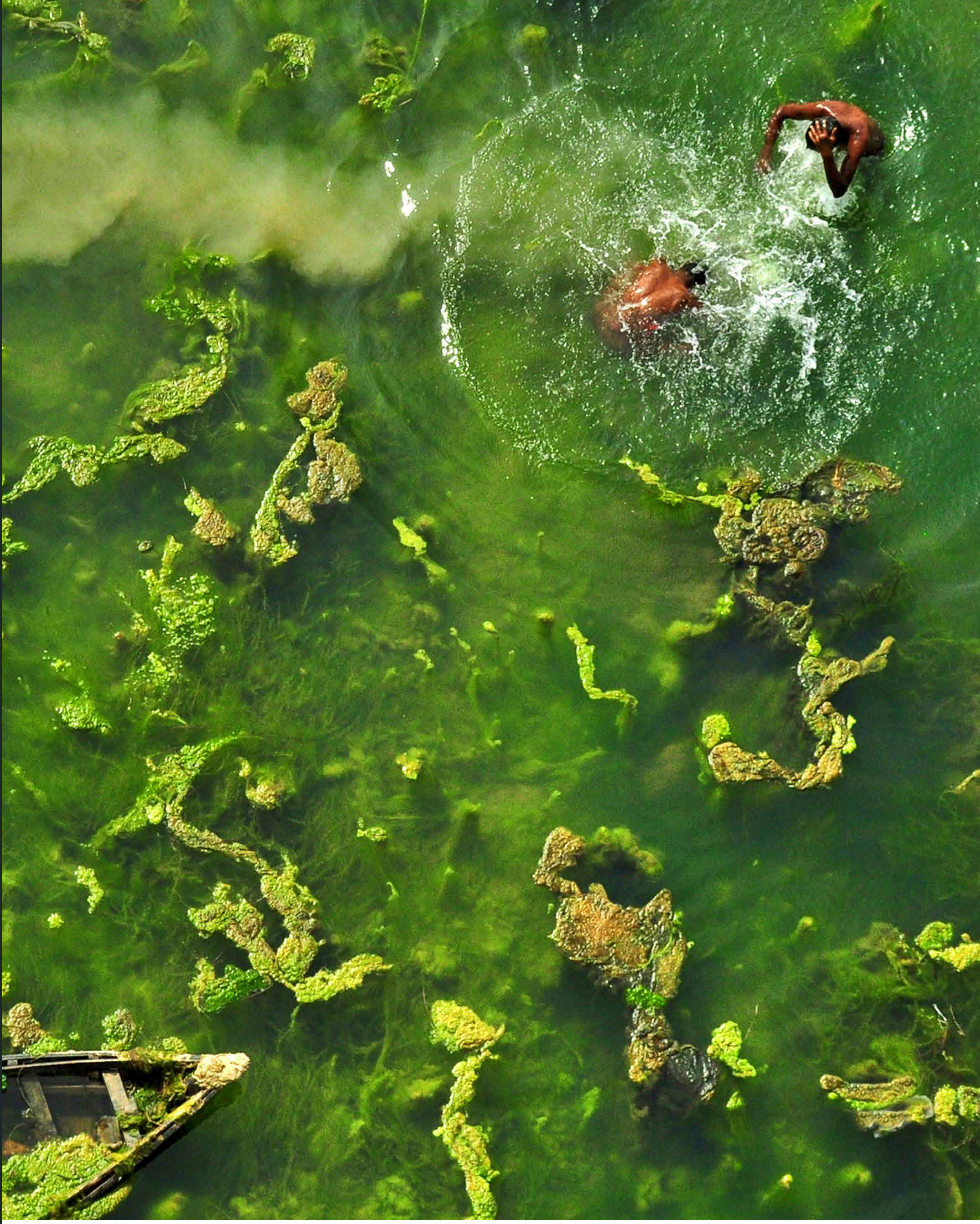

Our Phosphorus Future

Towards global phosphorus sustainability

With support from the Natural Environment Research Council, The United Nations Environment Programme, The Global Environment Facility, through the International Nitrogen Management System project and the European Sustainable Phosphorus Platform.

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This report has been prepared by members of the Our Phosphorus Future network, as an output of the INMS project and a peer-reviewed contribution to the work of the GPNM. This Global Overview has been prepared as a scientifically independent process. This report represents the views of the authors and does not necessarily reflect the policies of the contributing organisations. As an overview, this report does not attempt to reach a consensus on all issues.

OUR PHOSPHORUS FUTURE

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gef INMS

European Sustainable Phosphorus Platform

UKRI Natural Environment Research Council

Global Partnership on Nutrient Management GPNM

Cover photo show two men bathing in a phosphorus polluted river in West Bengal India. Photograph taken by Apratim Pal: https://instagram.com/guycalledapratim?utm_medium=copy_link

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Foreword

We know from the UNEP International Resource Panel's work, which I'm co-chairing with Izabella Teixeira, that the extraction and processing of natural resources drive all aspects of the triple planetary crisis of climate change, biodiversity loss, and pollution with health implications included.

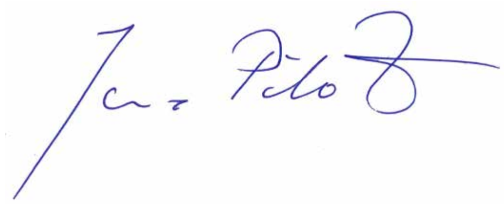
Nutrients – such as phosphorus, nitrogen, and potassium, are essential natural resources for life on this planet. Yet, in too high a quantity, they can cause profound environmental problems. Phosphorus presents a unique and complicated case. It is an essential driver of the global food system and as such, is enshrined within global scale biogeochemical, trade, and policy arenas. Humans have altered the global phosphorus cycle to meet food demands, primarily through the application of phosphorus in fertilisers to increase crop yield and support livestock production.

The planetary boundary for biogeochemical flows of phosphorus is one of the furthest transgressed. In some areas of the world, phosphorus has been added to soils in excess, whereas, in others, soils remain phosphorus limited. In the former, losses of phosphorus through the food system to lakes, rivers, wetlands, and coastal ecosystems, can breach their natural assimilative capacity, for example, driving harmful algal blooms, biodiversity loss, and threatening drinking water supplies. In the latter, the challenge is to ensure equitable access to farmers of phosphorus fertilisers to meet current and future food demands, whilst minimising environmental impacts.

This report demonstrates that the scientific knowledge is available with which to plan for a more sustainable phosphorus future. It presents a timely synthesis across many related fields to set out clear opportunities for increasing global phosphorus sustainability, viewing waste reduction and optimised recycling of phosphorus in the context of developing a global-scale circular economy for phosphorus.

This report represents a contribution to the work of UNEP's Global Partnership on Nutrient Management (GPNM), which was launched during the 17th session of the UN Commission on Sustainable Development in 2009 to link governments, policymakers, industry, science community, civil society organizations and UN agencies, with UNEP providing the Secretariat. More recently, the 4th session of the UN Environment Assembly in March 2019 adopted a landmark resolution on Sustainable Nitrogen Management, which was followed by the Colombo Declaration in October 2019 marking the launch of a UN Global Campaign on Sustainable Nitrogen Management. It is my hope that the information contained in this report 'Our Phosphorus Future' will help raise the profile of phosphorus, informing cross-sector discussions and collaborations, ultimately catalysing change toward sustainable phosphorus management. We must embrace such strategies if we are to achieve cleaner waters, healthier people and improve the well-being of many.

For the *Future We Want* there is no other way than to make our economies and society more resilient and better prepared for the growing challenges we already face. This is calling for the system change approach: minimising trade-offs and future lock-ins and maximising co-benefits and synergies among all our efforts. Focusing only on cleaning the current production systems will unfortunately not be enough and we must also enter the untapped territories of the needed deep system transformation. If we want to avoid the extinction of elephants in nature, we should take care that there are no more elephants in the room.



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Vice-chair of UNEP's Global Partnership on Nutrient Management.

Preface

Historically, nutrient management has been fragmented by natural, legal and administrative boundaries. It is widely recognised that land-based human activities significantly impact freshwater and the coastal environment. The Global Partnership on Nutrient Management (GPNM), formed in 2009, is a partnership of governments, scientists, policymakers, the private sector, NGOs and international organisations. The GPNM recognises the need for strategic advocacy and cooperation at the global level to communicate and inform dialogue not only on the complexity of the nutrient challenge but also to develop new opportunities for cost-effective policy and investment interventions by countries.

The Our Phosphorus Future report focuses on the challenges and solutions to deliver global phosphorus sustainability, and specifically on how poor phosphorus management impacts food security and the quality and availability of freshwater and coastal resources. This report identifies the numerous pathways through which land-based activities generate deleterious impacts due to phosphorus mismanagement, acknowledging that they can differ, depending on the location, type, condition and resilience of the local ecosystems. It also identifies opportunities for the circular economy of changing phosphorus management. The report identifies the potential for economic benefits of improving phosphorus sustainability, illustrated by estimates of the consequences of meeting a global aspirational goal to make a 50% reduction in global phosphorus pollution and a 50% increase in the recycling of phosphorus lost in residues/wastes, by 2050. We thank the lead authors and their team for their dedicated efforts to draw together an evidence base that demonstrates the need for enhanced governance coordination on phosphorus. As the report advocates, future governance systems should not be constrained by existing boundaries which often disconnect causes from effects. Instead, greater emphasis should be placed on safeguarding our natural resources, advancing the Sustainable Development Goals of the Agenda 2030, and breaking away from current unsustainable resource use patterns.

Executive summary

Key Messages

Unsustainable phosphorus use is at the heart of many societal challenges. Unsustainable phosphorus use affects food and water security, freshwater biodiversity and human health. Increasing demand for food to support a growing global population continues to drive increases in phosphorus inputs to the food–system, as well as losses from land-based sources to freshwater and coastal ecosystems. These losses cause ecological degradation through the proliferation of harmful algal blooms in fresh waters, contributing to alarmingly high rates of biodiversity decline, economic losses associated with clean-up, and large-scale human health risks from contaminated drinking water supplies. The pace of species extinction, climate change and the growing number of extreme weather events, combined with population growth and the economic impact of COVID-19, have further strengthened the need to invest in phosphorus sustainability.

Challenges

The global anthropogenic phosphorus cycle is unsustainable. Phosphate rock is a non-substitutable, non-renewable natural resource, essential for fertilisers and animal feeds, and so for global food security. Phosphorus is also important in smaller quantities in industrial applications. Phosphorus emissions throughout agriculture, food and sewage systems are predicted to increase under global business-as-usual scenarios. The sources of emissions differ significantly between regions. Whilst agricultural systems vary, poor phosphorus management is widespread. Estimated losses of phosphorus from agriculture to surface waters account for about 34% of global fertiliser use (~5 Mt phosphorus year^{-1}) representing 56% of all terrestrial inputs to surface waters (see Section 5.5). In some regions, including parts of Africa and India, wastewaters are the dominant source of phosphorus emissions with wastes often discharged directly to rivers with no treatment. Globally, ~80% of all wastewaters are discharged without treatment (in low-income countries ~8% are treated, in high-income countries ~70% are treated) (see Section 5.5).

Affordable access to sustainable phosphorus sources is imperative to ensure food provision for all and to protect the livelihoods of smallholder and marginal farmers. Currently, 1 in 7 farmers cannot afford sufficient fertilisers to maintain fertile soils, impacting their ability to produce food (see Section 3.3). Without change, insufficient phosphorus fertiliser use in many parts of Africa will likely lead to crop yield reductions of nearly 30% by 2050 (see Section 3.3). Phosphorus additions to increase aquaculture yield are a growing and direct pollution threat to this food system and the freshwater and coastal ecosystems it relies upon. Almost half of the world's population rely on fish for 20% of their protein intake making aquaculture a critical global industry, with an annual turnover of US$ 160 billion (see Section 5.5). Significant improvements in phosphorus use efficiency can be made across these critical food provision sectors.

High dependency on imported phosphate rock and/or mineral phosphorus fertilisers can contribute to national food system vulnerability. Agriculture is entrenched in its reliance on mineral phosphorus fertiliser; 85% of phosphates produced for the market are processed to make mineral fertilisers and 10% are used to make animal feed supplements (see Section 2.1). Five countries hold 85% of known phosphate rock reserves, with 70% found within Morocco and Western Sahara. Geological depletion of phosphate rock is not an immediate threat, with a current global estimate of over 300 years of phosphate reserves (see Section 2.1). However, geopolitical, institutional and economic factors can impact phosphorus access domestically. Improving the efficient use of phosphorus in agriculture (see Section 4.6) and shifting reliance away from mined phosphorus sources by increasing phosphorus recycling (see Sections 6.4 and 7.4) could help to reduce supply risk, at least at a national scale.

Aquatic ecosystems are under severe stress from phosphorus pollution. The rate of biodiversity loss in fresh waters is higher than in any other planetary domain, and nutrient pollution is a key stressor (see Section 5.1). Globally, phosphorus losses from land to fresh waters have doubled in the last century and continue to increase, contributing to toxic algal blooms, biodiversity loss, and threatening human and environmental health. Climate change is expected to increase the severity of these impacts, whilst the release of greenhouse gases from phosphorus enriched lakes exacerbates climate change (see Section 5.4.2). Reducing nutrient emissions to fresh waters has been listed as a priority for most countries to redress water quality degradation, in line with the United Nations Sustainable Development Goal (SDG) 6.3.2.

Restoring ecosystems is notoriously difficult and carries unacceptable clean-up costs. The data required to assess the money spent to address the impacts of unsustainable phosphorus use remains limited. Of the few studies published, it has been estimated that eutrophication costs the US economy US$2.2 billion annually (see Section 5.5). In the UK, similar assessments indicate losses will increase from around £173 million ($220 million) in 2018 to over £400 million (>US$500 million) by 2080 as a result of climate warming alone (see Section 5.5). The present report estimates the global cost to implement catchment management to intercept phosphorus losses to fresh waters from anthropogenic sources could reach about US$265 billion year^{-1} (see Section 5.5). Despite this estimate, global baseline data on phosphorus emissions and impacts are limited. It is important to raise awareness of ecosystems under threat and to work across governments to ensure long-term ecosystem integrity through preventative management programmes.

Solutions

Phosphorus emissions from land-based sources represent an opportunity to reduce global reliance on mined phosphate rock, whilst relieving stress on freshwater and coastal ecosystems. A move towards a circular phosphorus economy stands to increase the resilience of national scale food systems.

A global commitment to recycling nutrients in wastes and residues is needed. Recycling phosphorus-rich organic residues and manures is critical for phosphorus sustainability. Multiple strategies exist to improve the recycling of phosphorus in manures, abattoir residues, food processing and domestic wastes, sewage derived biosolids and wastewaters (see Sections 6.4 and 7.4). Beyond agronomic benefits, the win-wins are numerous, with benefits to society, environment, economy and business growth. Phosphorus recovery processes provide the opportunity to produce contaminant free, high purity phosphorus products that may substitute for mined phosphorus (see Section 7.4). To increase phosphorus recycling significantly, education, awareness-raising, investment in technology and infrastructure, and policy support are urgently needed. A goal for fertiliser products to contain a minimum of 20% recycled phosphorus by 2030, could set a benchmark that demonstrates green commitment across the fertiliser industry (see Section 2.6).

Reducing excessive consumption of animal products (e.g. meat, dairy, and eggs) and decreasing food waste will significantly reduce phosphorus losses from the food system (see Section 8.3). Consuming products grown with good on-farm nutrient management practices, including phosphorus recycling can further reduce losses. Over the last 60 years, 38% of the increased use of mineral phosphorus fertilisers can be attributed to global diet changes (see Section 8.1). This increase is predominantly related to increased consumption of animal products, especially in wealthier countries, where per-capita consumption is often higher than is recommended for healthy diets.

An international framework is needed to address the lack of accurate baseline data on many of the major phosphorus flows and stocks at national, regional, and global scales. Assessments are needed that quantify the extent of eutrophication, the costs of impacts and of necessary mitigation actions to accelerate efforts at ecosystem restoration and to prevent future damage.

Ten key actions

Ten key actions across sectors are proposed to improve sustainable phosphorus management globally (see Section 9.2). Among these actions, priorities and preferred solutions can be expected to differ nationally and between regions.

1. Increase the use of recycled phosphorus in fertiliser and other chemical industries, as an alternative or supplement to phosphate rock.
2. Optimise phosphorus inputs to agricultural soils and maximise crop uptake to minimise losses.
3. Optimise animal diets and the use of supplements to reduce phosphorus excretion.
4. Increase appropriate application of manures, other phosphorus-rich

residues, and recycled fertilisers to soils, to complement appropriate mineral fertiliser use.

5. Improve global reporting and assessments of phosphorus emissions and their impacts on freshwater and coastal ecosystems.
6. Implement integrated approaches for freshwater and coastal ecosystem restoration and protection at catchment, national and transboundary scales.
7. Implement national to global strategies to increase recovery and recycling of phosphorus from solid and liquid residue streams.
8. Ensure sufficient access to affordable phosphorus fertilisers (mineral, organic and recycled) for all farmers.
9. Promote a global shift to healthy and nutritious diets with low phosphorus footprints.
10. Reduce the amounts of phosphorus lost as food waste in food processing, retail, and domestic consumption.

Towards a Sustainable Phosphorus Future

Looking to the future, significant investment aligned with increased public awareness and political support is needed to implement the solutions outlined in this report. A decade has passed since the global anthropogenic flow of phosphorus was assessed as having crossed the planetary boundary.[i] Yet, despite clear opportunities to move towards more sustainable phosphorus use, there remains a lack of direction in relevant food and environmental policy to support such a transition. Intergovernmental coordination is urgently needed to address this issue (see Section 9.4). Multiple benefits are associated with sustainable phosphorus use, including:

- Improved sanitation, essential for health and the environment.
- Healthier diets for some individuals.
- New employment opportunities through the nutrient circular economy.
- Coherence with sustainable management of other nutrients including nitrogen, carbon and potassium.
- Return of organic carbon to soils, contributing to soil fertility and climate resilience.
- Reduction in greenhouse gas emissions including carbon dioxide and methane, and potential synergies with nitrous oxide.
- Reduced national dependency on the limited regions with phosphate rock reserves.
- Reduced mobilisation of contaminants contained in some phosphate rock reserves.
- Increased biodiversity and socioeconomic benefits associated with ecosystem recovery.

A transition towards more sustainable phosphorus use will help countries contribute to their commitments to multiple UN-SDGs, that include:

[i] A level of human interference in the global phosphorus cycle that results in potentially irreversible environmental damage (see Section 2.3).

SDG 1 – No poverty and SDG 2 – Zero Hunger, through the development of business growth within the circular economy, risk reduction to sectors (and employees) reliant on healthy aquatic ecosystems and reduction in poverty-related malnutrition through the protection and provision of livelihoods.

SDG 3 – Good Health and Well-Being, by reducing the risk of harmful algal blooms, and a reduction in illnesses from hazardous water pollution (e.g., cyanotoxins produced during harmful algal blooms).

SDG 6 – Clean Water and Sanitation, through a reduced risk to drinking water supplies resulting from improvement to water resources impacted by phosphorus pollution and the protection and restoration of aquatic ecosystems, and improved sanitation where phosphorus recovery drives infrastructure investment.

SDG 12 – Responsible consumption and production, through improved sustainable management and efficient use of natural phosphorus resources, and improving environmentally sound management of chemicals (e.g. fertilisers) and all wastes throughout their life cycle.

SDG 13 – Climate Action, through reduced contributions to greenhouse gas emissions from phosphorus polluted ecosystems.

SDG 14 – Life Below Water, through the sustainable management and protection of marine and coastal ecosystems to avoid significant adverse impacts, including by strengthening their resilience.

SDG 17 – Partnerships, through improved sustainable phosphorus partnerships reliant on the development, transfer, dissemination, and diffusion of environmentally sound technologies to all countries.

An aspirational goal for phosphorus is proposed

The following goal is identified as an interim focus for 2050, which would together represent a major step on the pathway to a sustainable phosphorus future.

The OPF '50:50:50' Goal calls for a 50% reduction in global phosphorus pollution and a 50% increase in the recycling of phosphorus lost in residues and wastes, by 2050.

Globally achieving this target would deliver benefits across all seven of the OPF pillars. Key benefits of achieving the '50:50:50' goal for each OPF pillar, are listed below (see Section 9.5).

<u>Benefits to 'Phosphorus Access'</u>: Achieving the '50:50:50' goal could return an additional 8.5 MT of recycled phosphorus to farms each year, supporting food production and food system resilience.

<u>Benefits to 'Food Security'</u>: Achieving the '50:50:50' goal could create a food system that would provide enough phosphorus to sustain over 4 times the current population.

<u>Benefits to 'Agriculture and food production'</u>: Achieving the '50:50:50' goal could save the global farming community almost $US4 billion in annual mineral phosphorus fertiliser costs needed to replace losses.

<u>Benefits to 'Water Quality'</u>: Achieving the '50:50:50' goal could significantly reduce the impacts of eutrophication, cutting the need for adaptation costs by over US$250 billion year^{-1}, with socio-economic benefits of restored ecosystems, including greater biodiversity and growth of ecotourism.

Benefits for 'Recycling': Achieving the '50:50:50' goal could support a transition to a circular economy for the phosphorus cycle; decoupling economic growth from the consumption of finite phosphate rock resources.

Benefits for 'Recovery': Achieving the '50:50:50' goal could develop sustainable business opportunities, accelerating and supporting new jobs through emerging green economy sectors.

Benefits to 'Sustainable Consumption': Achieving the '50:50:50' goal could provide consumers with better access to foods produced in phosphorus sustainable ways, allowing consumers to better support a transition to a sustainable phosphorus future, sustainable city living and post COVID-19 'Green Recovery'.

The '50:50:50' goal aligns with several aspirational goals that have also called for reductions in nutrient losses in recent years. These include:

- The United Nations Environment Programme (UNEP) Colombo Declaration which calls for the halving of nitrogen (N) waste by 2030.
- The working group of the Post-2020 Global Biodiversity Framework which proposed to reduce pollution from excess nutrients by 50% by 2030.
- The Farm to Fork strategy underpinning the European Green Deal, which calls for actions to reduce nutrient losses by at least 50% and to reduce fertiliser use by at least 20% by 2030 (see Section 9.5).

If the world is to meet climate change, biodiversity, and food security targets, and avoid building costs of predicted phosphorus impacts, positive action on phosphorus management is essential. The present report calls for the establishment of an intergovernmental coordination mechanism to catalyse integrated action on phosphorus sustainability (see Figure 9.1). This should be supported by an international framework to consolidate the collective knowledge, quantify the economic and societal benefits of improvements in phosphorus management and establish targets for time-bound improvements.

The report identifies a clear opportunity to raise awareness of the need for sustainable phosphorus management through the United Nations Environment Assembly (UNEA) and calls for a UNEA resolution on sustainable phosphorus management or an equivalent global commitment to act.

01

Our Phosphorus Future - an introduction

Authors: Bryan M. Spears, Mark A. Sutton, Kate V. Heal, Dave S. Reay, Will J. Brownlie

Left: Lake Braies in the Prags Dolomites in South Tyrol, Italy. Ecotourism in this area is reliant on healthy aquatic ecosystems and high water quality. Photograph taken by Geoffrey Lucas on www.unsplash.com - www.geoffphotography.com/

The 'Our Phosphorus Future' project (OPF) responds to the critical need to provide direction from the global phosphorus scientific community to progress sustainable phosphorus use. The OPF project ran from 2017-2021. During this time over 100 scientists and industry experts came together to develop this report. The report identifies the priority issues, possible solutions and the capacity to address phosphorus sustainability from local to global scales.

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1.1 What is the Our Phosphorus Future project?

The 'Our Phosphorus Future' (OPF) project is a response from the scientific community to the need for direction on sustainable phosphorus (P) use (Focus Box 1.1). The OPF report identifies the priority issues, possible solutions and the potential to improve P sustainability from local to global scales. At the same time, it aims to prime the international scientific, practitioner and policy communities to co-develop the next steps towards a durable international process of scientific support for P policy. A principal aim has been to consolidate scientific evidence and use it to raise awareness of the need to improve P sustainability.

From 2017-2021, over 100 scientists and industry experts from around the world have combined efforts to develop the OPF Report. The project has been delivered through a partnership between the UK Centre for Ecology & Hydrology (UKCEH) and the University of Edinburgh, UK, and with funding from the UK Natural Environment Research Council (NERC), the European Sustainable Phosphorus Platform (ESPP), the United Nations Environment Programme (UNEP) and the Global Environment Facility (GEF) through the GEF/UNEP 'Towards the International Nitrogen Management System' (INMS) project.

This report is not designed to produce binding recommendations, but instead to inform discussions and raise awareness through appropriate fora. International teams of authors were invited to produce stand-alone chapters to cover the central components of P sustainability (Chapters 2-8). These components, referred to from here on as the 'OPF Pillars', were proposed by the Project Management Group and developed further by the co-authors of the chapters (Figure 1.1). Chapter 9 synthesises the key challenges and solutions identified across all OPF Pillars, integrating these across related sustainable nutrient management initiatives, from which it proposes a road map for improving future integration, targeted at encouraging wider community discussion.

This report represents hundreds of hours of scientific discussion and peer review and is underpinned by >2000 peer-reviewed publications and reports spanning more than 300 years of scientific research. The report has undergone an extensive review process (Figure 1.2), supported by more than 40 referees from both academia and industry bodies.

THE OPF PILLARS	CHAPTERS IN THE OPF REPORT
PHOSPHORUS ACCESS	**CHAPTER 2.** PHOSPHATE ROCK, RESERVES AND USES
FOOD SECURITY	**CHAPTER 3.** TRANSFORMING THE FOOD SYSTEM – IMPLICATIONS FOR PHOSPHORUS
AGRICULTURE AND FOOD PRODUCTION	**CHAPTER 4.** OPPORTUNITIES FOR BETTER PHOSPHORUS USE IN AGRICULTURE
WATER QUALITY	**CHAPTER 5.** PHOSPHORUS AND WATER QUALITY
PHOSPHORUS RECYCLING	**CHAPTER 6.** OPPORTUNITIES TO RECYCLE PHOSPHORUS-RICH ORGANIC MATERIALS
PHOSPHORUS RECOVERY	**CHAPTER 7.** OPPORTUNITIES TO RECOVER PHOSPHORUS FROM RESIDUE STREAMS
CONSUMPTION	**CHAPTER 8.** CONSUMPTION – THE MISSING LINK TOWARDS PHOSPHORUS SECURITY

Figure 1.1 The Seven Pillars of the OPF Report, which underpin the central components of phosphorus sustainability and corresponding chapter titles.

THE REVIEW PROCESS OF CHAPTERS 2 - 8

- Authors and editors (i.e. Project Management Group) co-developed 2-page summaries outlining proposed chapter content.
- Authorship prepared 1st draft of full chapters.
- **1st draft reviewed by editors.**
- Authorship prepared 2nd draft of chapters.
- **2nd draft reviewed by the Scientific Advisory Committee.**
- Authorship prepared 3rd draft.
- **3rd drafts reviewed by Stakeholder Review Panel.**
- Authors addressed comments (over 80 pages of comments received) – this took over a year.
- The editors reviewed final chapters to ensure consistency and remove excess duplication with other chapters.
- The chapter text was sent for final approval and sign-off from all authors.
- Production team converted final text into chapter proofs, visual summaries and videos.
- Proofs, visual summaries and videos were shared for final sign-off from all authors.

Figure 1.2 The OPF review process. The review process involved three stages of review, with reviewers selected from academia, government and industry.

Focus Box 1.1 - Statements on the importance of sustainable phosphorus management in delivering global scale ambitions.

"A new global effort is needed to address 'The Nutrient Nexus', where reduced nutrient losses and improved nutrient use efficiency across all sectors simultaneously provide the foundation for a Greener Economy to produce more food and energy while reducing environmental pollution."

Sutton et al. (2013). Our Nutrient World.

"The accelerated use of nitrogen and phosphorus is at the centre of a complex web of development benefits and environmental problems. They are key to crop production and half of the world's food security is dependent on nitrogen and phosphorus fertiliser use. But excess nutrients from fertilizers, fossil fuel burning, and wastewater from humans, livestock, aquaculture and industry lead to air, water, soil and marine pollution, with loss of biodiversity and fish, destruction of ozone and additional global warming potential. The problems will intensify as the demand for food and bio-fuels increase, and growing urban populations produce more wastewater. This will be at a growing economic cost to countries in the undermining of ecosystems, notably in the coastal zone, and the services and jobs they provide."

Global Partnership on Nutrient Management (2011). The Nutrient Challenge.

"As the global population grows, the enormous problem of producing sufficient food in a sustainable manner will only intensify. Technological innovations and sustainable food production systems can decrease the sector's contribution to climate change, land-use change and ocean degradation; reduce environmentally damaging inputs and waste; improve production system resilience, through methods such as precision agriculture, integrated pest management and molecular breeding techniques; and are likely to have a positive economic impact, including the creation of jobs."

Dasgupta (2021) The Economics of Biodiversity: The Dasgupta Review.

"Action is needed: water quality needs to be politically prioritized, and it should be treated as an urgent concern for public health, the economy, and ecosystems. The findings from this report show that long-term costs have been underestimated and underappreciated. The threats that poor water quality presents are largely imperceptible, and as a result, policy inaction and procrastination are often convenient responses to an invisible problem. But this means that populations are subjected to hazards without their knowledge or their consent. With water scarcity expected to increase as populations grow and the climate changes, the world cannot afford to waste and contaminate its precious water resources."

Damania et al. (2019). Quality Unknown: The Invisible Water Crisis, World Bank.

"The fertilizer industry is aware of the role of fertilizers in nutrient losses to water (and more broadly to the environment) and is actively engaged in reducing such losses in partnership with farmers, their advisors and other relevant stakeholders. Nutrient losses can be minimized when best practices in farm and, more specifically, soil, water and nutrient management are applied."

International Fertiliser Association (2018). AGENDA 2030 Helping to Transform our World.

"The release of nutrients from agriculture and untreated wastewater poses the most widespread threat to environmental water quality globally. An in-depth analysis of submissions from countries that supplied parameter-level data showed that nitrogen and phosphorus failed to meet their

targets more often than the other water quality parameters of Level 1 reporting. This means that for these countries, and quite likely for most countries, reducing nutrient release and transport will have the greatest positive impact on water quality."

United Nations Environment Programme (2021a). Progress on ambient water quality. Tracking SDG 6 series: global indicator 6.3.2 updates and acceleration needs.

"Sustainable food systems work with nature, adapt to a warming world, minimize environmental impacts, eliminate hunger and improve human health. Sustainable food production is vital to protecting nature and human well-being. It can be achieved through a range of overlapping approaches, including conservation agriculture, organic farming, agroecology, integrated pest and nutrient management, soil and water conservation, conservation aquaculture, sustainable grazing, agroforestry, silvopastoral systems, irrigation management, small or patch systems and practices to improve animal welfare. Sustainable agriculture requires a reduction in nitrogen and phosphorus imbalances to reduce pollution of freshwater, groundwater and coastal zones."

United Nations Environment Programme (2021b). Making Peace with Nature: A scientific blueprint to tackle the climate, biodiversity and pollution emergencies.

"It is clear from this study that land-based activities generate multiple natural and non-natural stressors that impact the condition of coastal resources. Particularly impactful stressors were increasing concentrations of sediment (such as those from infrastructure development or poor land management), increasing concentrations of persistent toxins (including from mining runoff), increasing concentrations of plastic (for instance, from poor industrial production processes) and increasing concentrations of nitrogen and phosphorus (from, inter alia, poor agricultural practices). Agriculture, ports/harbours and aquaculture were the land-based activities with the greatest cumulative impacts on coastal resources, and should therefore be governance priorities."

IRP (2021). Governing Coastal Resources: Implications for a Sustainable Blue Economy.

"Phosphorus is essential for food production, but its global supply is limited. Better insight is needed into the availability of this non-renewable resource and the environmental consequences associated with its use. Optimizing agricultural practices while exploring innovative approaches to sustainable use can reduce environmental pressures and enhance the long-term supply of this important plant nutrient."

Syers et al. (2011) UNEP Year Book 2011: Emerging issues in our global environment, United Nations Environment Programme

"Noting with concern that excessive levels of nutrients, in particular reactive nitrogen and phosphorus, have significant impacts on species composition in terrestrial, freshwater and coastal ecosystems, with cascading effects on biodiversity, soil, water and air quality, ecosystem function and human well-being"

UNEA. 2022. UNEP/EA.5/L.12/Res.1. Draft resolution on sustainable nitrogen management.

1.2 A brief history of phosphorus

1.2.1 Discovery of phosphorus

Hennig Brand, in 1669, Hamburg, is widely credited in contemporary popular texts with the discovery of the first known element, phosphorus. The discovery, often romanticised, is depicted by an alchemist at his craft in search of the philosopher's stone (e.g. Focus Box 1.2). However, references to the discovery in modern literature suggest some contention in the details of the discovery, as addressed by Partington (1936). Kragh (2003) presents the case for Brand based on the conditions of credible discovery, where a substance is confirmed as being elemental and where details of the element are presented publicly for scrutiny. Indeed, it appears that Brand did neither, but instead disclosed the details of his experiments to contemporaries, Johann Kunckel and Daniel Kraft. Homberg (1692) described Brand as an uneducated person in search of the philosophers' stone, who, in his desire for secrecy, did not fully disclose his experiments to contemporaries such as Kunckel. Homberg (1692) claimed that Kunckel rediscovered phosphorus in his own laboratories following Brand's death. Leibniz (1710) having worked directly with Brand to replicate the process under the patronage of John Frederick, Duke of Brunswick-Calenberg, concluded that Homberg's account "departs from matter of fact". Leibniz goes on to give his account, based on having worked extensively with Brand:

"The invention of the phosphorus happen'd thus; Brand had fallen on a certain chemical process, extant in a printed book, which taught how to prepare from urine a liquor fit to ripen a particle off silver into gold; And in labouring on this he found out his phosphorus. He had some acquaintance with Daniel Kraft, then of the council of commerce to the [George III] Elector of Saxony; and by his means with Kunkel, one of the said Prince's bed-chamber; but who under that character perform'd chemical processes. On persuading Brand that this arcanum might be sold to the Great at a high price; and offering him their assistance, they obtain'd the composition from him. And upon going from Dresden to Hamburg, they both saw and learn'd from him the process of the phosphorus. But Kunkel, upon his return home, had committed some mistake in the process, and for a long time could not hit upon the phosphorus; and he sent a letter of complain to Brand, which I have seen, and in which he bewails that the secret was not communicated to him sincerely enough; but Brand repenting that he had been so easy in the communication, delay'd the setting him to rights. Kunkel, in the mean time, after various trials, corrected the error of himself; whence he pretended to be the inventor; of this Brand bitterly complain'd."

The first scientific account on P (called Noctiluca; 'night light') appears to have been published by Kirchmaier (1676), a colleague of Kunckel who published further detailed reports in his *Collegium physico-chemicum experimentale* (Kunckel, 1716). Robert Boyle, having been introduced to the product of Brand's experiments by Kraft in 1677, is credited with publishing the first account on the preparation of P in 1680 (Boyle, 1680). Boyle's contributions are reviewed in detail by Partington (1936); in which Brand's discovery is noted as

1675. The extent to which these works were influenced by Brand's methodology remains uncertain. The method was later reviewed in detail by Hellot (1737). The early history of P appears to be obscured by both academic and economic competition, with private correspondences and public presentations repeating claims and counter-claims. Beyond the initial scientific curiosity of phosphorescence, which in the case of elemental P was more long-lasting than other phosphorescent substances, potentially economic applications included uses in artificial lighting, matches and medicinal cures.

1.2.2 Scientific and economic development of phosphorus use

From the late 18th to early 20th centuries, P remained the focus of academic and economic attention, a period during which 13 scientists are credited with Nobel prizes for their work relating to P (Farber, 1965). Important discoveries relevant here include the isolation of phosphoric acid from bones by Gahn in 1769, the procedure for which is attributed to Scheele (described by Nordenskiold, 1892). The confirmation of P as an element appears to have resulted from the work of French chemist Lavoisier (1777), contributing to the development of modern day elemental nomenclature (Guyton de Morveau et al., 1787), and to Lavosier's seminal work *Traité élémentaire de chimie* (Lavoisier, 1789). This work initiated almost a century of elemental discovery and uncovered the importance of reactive hydrogen in determining the polybasicity of acids, pioneered by the study of phosphoric acid (Graham, 1833). Finally, utilising the above, von Liebig proposed acid digestion as a means for enhancing solubilisation of bones (with sulfuric or nitric acid, although not the first to do so; Farber, 1965) for improved P fertilisation of soils for plant growth (Liebig, 1843). This ultimately led to a patent being filed by John Bennet Lawes at Rothamsted for the production of 'superphosphate'. von Liebig's recommendation also accelerated the use of guano as a fertiliser around this time, opening up new markets in its extraction and supply and leading to the birth of modern agri-chemistry.

However, it was not until 1888 when James Burgess Readman developed the electric arc furnace for producing the element from phosphate rock, that mineral P fertilisers began to be produced at an industrial scale (Readman, 1889). These latter developments were central to the agricultural intensification of the 20th and 21st centuries. Phosphate rock demand increased in the USA, alone, from 0.2 Mt to 19 Mt between 1880 and 1962 (Farber, 1965), primarily for the production of fertilisers. The 'phosphorus rush' of the 20th century had begun.

1.2.3 An emerging understanding of the importance of phosphorus in freshwater ecology

The Swiss scientist François Alphonse Forel proposed the term 'Limnology' to describe the study of oceanography in lakes. Forel pioneered the field through exploration and description of the physical, chemical, and ecological condition of Swiss lakes, principally Lake Geneva (Forel, 1892). The development of the field progressed through various strands, which fostered collaborative works as reviewed by Egerton (2014). For example, Forel, and colleagues, contributed to an important collection of text on the Flora and Fauna of Fresh Water, edited by the German limnologist Zacharias (Zacharias, 1891). Around the same time, an extensive survey of 562 Scottish lochs was led by Sir John Murray and Laurence Pullar, involving the services of 50 scientists and surveyors, describing bathymetry alongside chemical, biological and geological observations (Murray and Pullar, 1910). This included one of the first global assessments in Murray's 'the characteristics of lakes in general, and their distribution over the surface of the globe' (Murray, 1910). Coincidentally, Murray, through his role in organising the collections of the Challenger expedition, and realising the economic potential of phosphate-rich guano deposits on Christmas Island, worked to influence British annexation of the island and co-founded the Christmas Island Phosphate Company. Between 1899 and 1913, some 1.4 Mt of phosphate rock were mined from Christmas Island, representing about 2% of global production at that time (Bustyn, 1975). Some of the proceeds of this venture were used to support further scientific exploration in both oceanography and limnology (Bustyn, 1975).

These building blocks of limnology were to further the understanding of ecological theory in fresh waters. The German limnologist August Friedrich Thienemann recognised that species of benthic chironomid larvae varied with oxygen concentrations in lake bottom waters, and, in turn, with phytoplankton development in surface waters; concluding that the communities were linked (Thienemann, 1921). The Swedish limnologist Einar Naumann described the natural succession of lake plankton communities using the terms 'oligotrophic' and 'eutrophic', representing the continuum of communities from nutrient-poor to rich waters, respectively (Naumann, 1921). Together, Thienemann and Naumann worked to further develop these concepts and fostered collaborative efforts in the field, establishing the Internationale Vereinigung für Limnologie, now the International Society of Limnology, which celebrated its centenary in 2022. This period saw the establishment of major research stations, including, for example, the UK Freshwater Biological Association at Windermere (1929) and the Plöner See Research Station (1891), now the Max Planck Institute for Evolutionary Biology (Germany), and others. An understanding of biogeochemistry in lakes and of interactions across food-webs was subsequently developed by Hutchinson, demonstrating, for example, rapid uptake of radio-isotope labelled P by phytoplankton and confirming that the supply of this nutrient is critical in determining ecological processes at the ecosystem scale (Hutchinson and Bowen, 1947; Hutchinson, 1957). Just as in agriculture, P was key to the growth of plants in freshwater ecosystems.

Shortly following World War II, attention was drawn to the effects of urban development on increasing P discharges to fresh waters from wastewater (Hasler, 1947). This resulted in a surge in scientific meetings and reports on cultural eutrophication, including notable case studies of water quality deterioration and increasing algal blooms in lakes of northern America (e.g., Lake Washington; Edmondson, 1961) and Europe (e.g., Lake Norrviken; Rodhe, 1948; Ahlgren, 1967). These events attracted international attention. For example, the historical development and global extent of cultural eutrophication and its causes were discussed in a special meeting of the Limnological Society of America on "Fertilization of Aquatic Areas," Boston, 1946, and have been reviewed in several contemporary texts (e.g., Wetzel, 2001; O'Sullivan and Reynolds, 2005).

The discoveries that followed set the paradigm for the present-day scientific response to cultural eutrophication (Chapter 5). For example, Vollenweider, (1968) examined nutrient loading to lakes from agriculture and other terrestrial sources. In doing so, he proposed modelling approaches with which to define P load reduction targets in line with water quality and ecological responses. Schindler (1974) and colleagues at the Canadian Experimental Lakes Area conducted whole-lake experiments to disentangle the processes operating during eutrophication, confirming P reduction to be a primary aim of eutrophication management. Between 1964 and 1974 the International Biological Programme of the International Council of Scientific Unions (ICSU), through its subcommittee on the Productivity of Freshwater Communities, coordinated a long term global monitoring programme to establish the 'productivity of biological resources, human adaptability to environmental change, and environmental change itself' (Burgis and Dunn, 2012).

Some of these programmes continue to date, providing essential insights into lake ecosystem responses to nutrients and other pressures. Lakes act as sentinel ecosystems with which to detect the ecological effects of long-term environmental change, including cultural eutrophication and ecological recovery following the reduction of nutrient emissions (e.g. Jeppesen et al., 2005; McCrackin et al., 2017). In some countries, large scale and coordinated monitoring programmes have been initiated in recent decades to provide large scale assessments of the impacts of nutrient emissions in both freshwater and coastal ecosystems (Chapter 5). The evidence produced from such programmes continues to underpin the development of lake basin and national scale environmental policies and directives (e.g., the Federal Water Pollution Control Act, USA, 1942; latterly the Clean Water Act, 1972). These have focussed primarily on the treatment or diversion of wastewater discharges and agricultural emissions (Chapter 5). As the United Nations Environment Programme works to support countries in developing monitoring and assessment programmes, designed to protect freshwater and coastal ecosystems, it is clear that much remains to be achieved in the coming decade on nutrient emissions and impacts to inform policy responses (Focus Box 1.1; UNEP, 2021a). Indeed, the Alliance for Freshwater Life (Darwall et al., 2018), in its recent call for a more coordinated response to the decline of freshwater biodiversity, argues that existing policies relevant to safeguarding freshwater ecosystems are failing due to a lack of conviction and enforcement in implementation. The implementation of the UN Sustainable Development Goals and the Decade on Ecosystem Restoration are primed to address this call from the scientific community, providing an opportunity to forge a new direction for sustainable nutrient use with a focus on ecosystem recovery and protection.

In recent years, resolutions of the United Nations Environment Assembly have brought sustainable nutrient management (UNEP/EA5/L12/REV.1), water quality assessment and improvement (UNEP/EA.3/Res.10), and sustainable lake management (UNEP/EA5/L8/REV.1) into focus. For example, UNEP/EA.3/Res.10 on "Addressing water pollution to protect and restore water-related ecosystems" requested UNEP to develop a global water quality assessment in collaboration with UN-Water and relevant stakeholders by UNEA-5. In response, and in addition to the World Water Quality Assessment process, UNEP coordinated the formation of the World Water Quality Alliance (WWQA), an open community of practice, representing a voluntary and flexible multi-stakeholder network with a shared goal; to improve freshwater quality to achieve prosperity and sustainability. The WWQA includes a strong focus on assessing global nutrient impacts on freshwater ecosystems and has mobilised a working group dedicated to accelerating ecosystem restoration globally, the WWQA Ecosystems Work-stream. This group and others are working together to ensure that nutrient management and water quality improvement are at the heart of sustainable development plans.

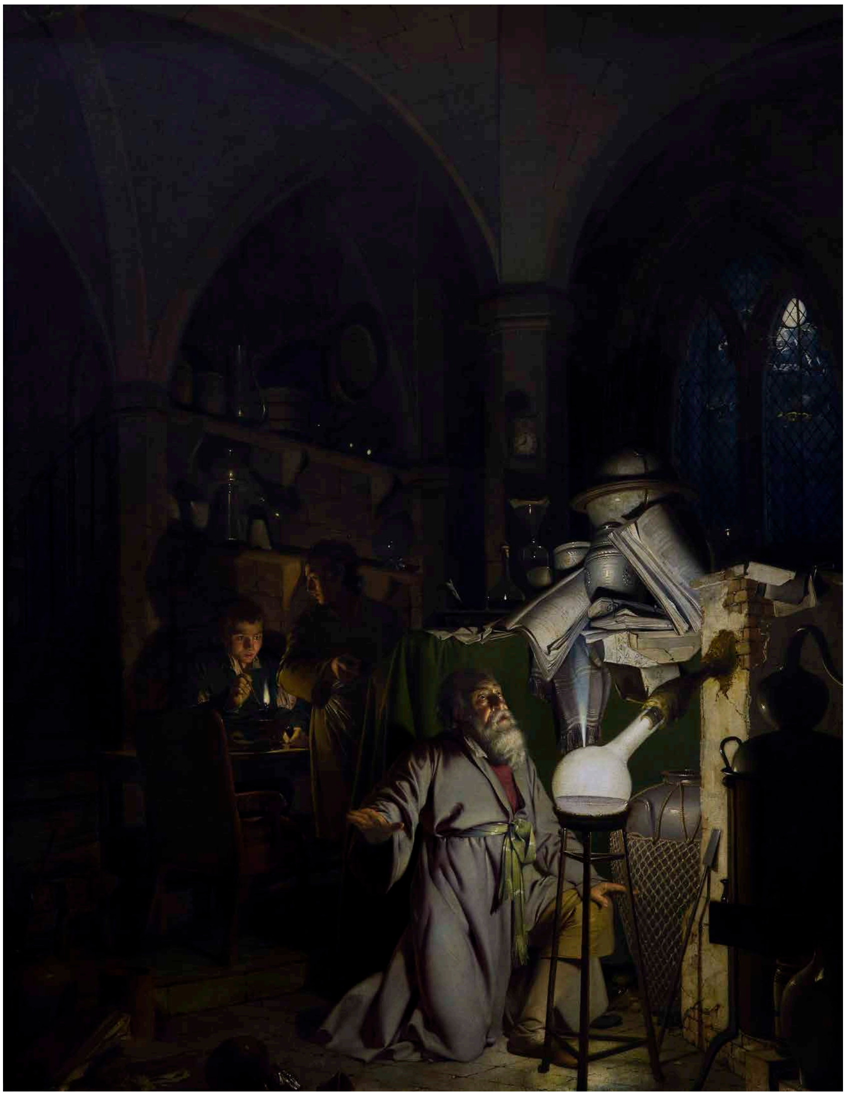

Figure 1.4 Painting Title: The Alchymist, in Search of the Philosopher's Stone, discovers Phosphorus, and prays for the successful conclusion of his operation, as was the custom of the ancient chymical astrologers (1771, by Joseph Wright), displayed in Derby Museum and Art Gallery, Derby, UK. Source: https://commons.wikimedia.org/wiki/File:Hennig_Brand.jpg#/media/File:JosephWright-Alchemist.jpg

Focus Box 1.1 - What is phosphorus and how was it discovered?

Authors: Jim Elser and Phil Haygarth, *passages excerpted from "Phosphorus Past and Future" By Jim Elser and Phil Haygarth (2021).*

What is phosphorus and how did we first learn about it?

"Did you know that there's ~0.62 kg (1.35 pounds) of phosphorus in your (average) body, right now, and that during your (average) lifetime you'll consume ~34 kg (75 pounds)? With too much phosphorus (or in the wrong form), you die. Without it, you die. With too much, humanity suffers. Without it, humanity falters. Phosphorus is vitally important and yet its role is often hidden and unappreciated."

But, what is phosphorus, actually?

"Remember that an atom is made up of protons (positively charged) and neutrons (no charge, hence their neutrality) in the nucleus and, buzzing around them, negatively charged electrons. … In the case of our hero phosphorus, there are 15 protons (and 15 neutrons), balanced by 15 electrons (Figure 1.3); phosphorus thus has an "atomic number" of 15. Figure 1 shows Bohr-like orbits, occupied by different numbers of electrons: two in the first, eight in the second, and five in the third, adding up to 15."

Our first knowledge of phosphorus came from urine….

"Lots of urine. The urine collected from dozens of beer-drinking German soldiers and from horses (presumably they abstained from beer). More than 5000 litres of it was brought, in big pails and all manner of containers, to the Hamburg workshop of a German named Hennig Brand (or Nicholas Brandt, depending on where you look). Today we call him an alchemist. In his day, the mid-17th century, he was a merchant and a high-tech entrepreneur, seeking health and wealth via the pursuit of the Philosopher's Stone. Back in those days, the Philosopher's Stone was a mythical material, knowledge of which was passed (inefficiently, it seems) from generation to generation and which was capable of transmuting lead into gold and even of extending human life (hence, it was thought, the impressive life spans reported for various Biblical figures). In the famous painting (Figure 1.4), we see Herr Brand, genuflecting before the glowing flask, his young helpers behind him looking a bit quizzical. The first chemical element had been purified. The date is 1669."

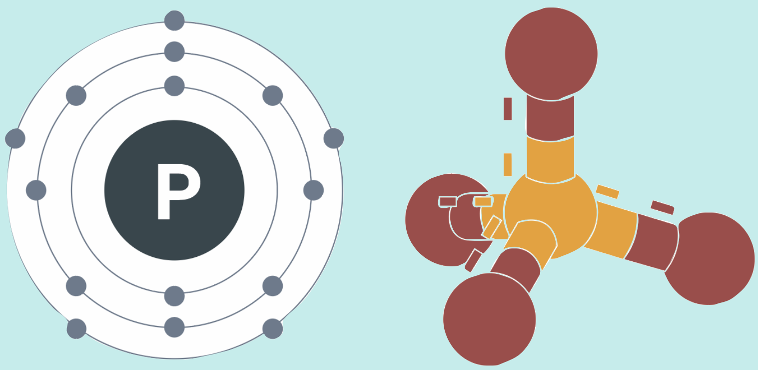

Figure 1.3 The structure of a phosphorus (P) atom (left). A 3D illusration (right) of a P atom bonded with four oxygen atoms to form phosphate (small dashed lines represent bonded pairs); this represents the most common chemical form of P in nature. Modified from Elser and Haygarth (2021).

1.3 Defining the global phosphorus sustainability challenge

1.3.1 Present concerns and future outlook

By the mid- to late-20th century, the sustainability of the global anthropogenic P cycle for the long-term provision of food and fresh water was brought sharply into focus. Stumm (1973) in his early assessment of the anthropogenic P cycle, concluded:

"By mining phosphorus in progressively increasing quantities, man disturbs the ecological balance and creates undesirable conditions in inland waters, estuaries and coastal marine waters… Our present agricultural practice of excessively fertilizing land needs to be re-examined; our present agricultural technology must not without modification be exported to tropical areas."

The suggestion here is that, if unchecked, unsustainable P use would create environmental conditions that are unacceptable for humanity. Rockström et al. (2009a; b) termed this concept, 'the Planetary Boundaries', and as Stumm correctly surmised, humanity was fast approaching the P boundary in the mid- to late-20th century.

Despite Stumm's warning, the global anthropogenic P cycle continued on its trajectory of increasing flows through the early 1970s (Chapters 2 and 3). The net result is that global P input to fresh waters had doubled by the end of the 20th century (Beusen et al., 2016). Carpenter and Bennett (2011) concluded that the planetary boundary for P had already been exceeded for freshwater eutrophication, but not yet for the extraction of phosphate rock. However, Carpenter and Bennett (2011) also raise the issue of balancing heterogeneity in P demand for food and emissions to fresh waters as a major future challenge.

The global outlook for anthropogenic P emissions to fresh waters is especially worrying. Phosphorus demand in the agricultural sector is predicted to double, again, by 2050 (from 2006 levels), further increasing risk of emissions to fresh waters (Mogollón et al., 2021). Phosphorus losses from food production for domestic consumption will impact directly on catchments that have been 'set-aside' for agriculture. Losses of P from wastewater to fresh waters could increase globally by up to 70%, by 2050 (van Puijenbroek et al., 2019). Yet, as is widely acknowledged (see Chapters 2-9), sustainable P management remains largely ignored in the food and environmental policy agendas of many countries, despite the social and economic burden it carries.

1.3.2 Geographic variation in phosphorus consumption

Arguably, progress on more sustainable P use has been hampered in recent decades by a fixation on whether or not depletion of the world's mineral P reserves (and resources) represents a risk to food security – the so-called Peak Phosphorus debate (Cordell et al., 2009; Van Vuuren et al., 2010; Syers et al., 2011; Scholz and Wellmer, 2013). Recent estimates indicate that phosphate rock reserve supply does not represent a near term risk for global food security, with a projected lifetime of P reserves of around 320 years (Jasinski, 2021).

However, national exposures to such risk are variable, especially where countries are reliant on either fertiliser exports as their economic foundation or fertiliser imports to maintain food security (Chapter 2). The fertiliser market is tight and subject to geopolitical and market tensions. The challenges of price spikes and export controls, such as experienced in 2008 (Chapter 3) and in 2021 are expected to continue.

Assuming business as usual for sustainable P management, Sutton et al., present two simple scenarios to frame the consequences of either P replete or limited supply chains, both of which entail significant societal challenges that must be addressed (see page 4 in Sutton et al., 2013).

- **Phosphorus limited scenario.** Insufficient P supplies lead to higher fertiliser prices, food prices and higher food system vulnerability, all of which will be exacerbated by increased human population and growing per capita demand for animal products. There is little economic capacity to address pollution in freshwater and coastal ecosystems.
- **Phosphorus replete scenario.** The continued supply of relatively cheap P fertilisers provides weak motivation to avoid losses to the environment, with the result that freshwater and coastal pollution problems are exacerbated through meeting increased demand for food to meet the growing population.

The geographic concentration of phosphate rock reserves risks impacting food security for countries and regions dependent on imported phosphate rock and/or fertilisers (e.g. Sub-Saharan Africa, India, the EU, Australia and Brazil) (Jasinski, 2021). Unsurprisingly, wealth is an important determinant of whether a nation's farmers have access to P fertilisers (Obersteiner et al., 2013). When describing P fertiliser consumption patterns, most countries can be classified in, or are transitioning between, one of three broad categories, each with different P sustainability issues:

- **Countries that consume too little phosphorus.** Typically less economically developed countries, e.g. some nations in Sub-Saharan Africa and parts of Asia. In these countries insufficient access to P fertiliser can constrain agricultural production, impacting food security. Often these countries have a growing population, increasing urbanisation and poor sanitation. This can create 'hotspots' of P loss from human wastes in and around cities, contributing to eutrophication.
- **Countries that are significantly increasing phosphorus consumption.** Typically large countries with emerging economies, e.g. Brazil, India, and China. In these countries increasing mineral P fertiliser use is contributing to rapid increases in agricultural output. However, low P use efficiency, and in some cases, insufficient sanitation, often cause substantial P losses resulting in increasing and/or significant eutrophication issues.

- **Countries levelling off or reducing phosphorus consumption.** Typically more economically developed countries, e.g. most nations in the EU, and the USA. In these countries, long-term high consumption of P fertiliser has fuelled agricultural sectors. However, typically high and/or improving P use efficiency combined with improved access to legacy P stores in soils is allowing a levelling off or reduction in P fertiliser use. Good sanitation mitigates some P losses from human wastes in comparison to less economically developed countries. However, historic poor P management has left widespread chronic eutrophication issues, representing a financial, environmental and human health burden.

1.4 Towards a sustainable phosphorus future

As the global anthropogenic nitrogen and carbon cycles are being transformed towards more sustainable futures, the 'business as usual' outlook for P remains locked-in to the outdated assumption that 'to sustain food production and economic growth, we must bear the cost of environmental degradation, today'. Yet, as laid out in detail in subsequent chapters, even though the evidence exists to support a more sustainable future for P, it remains out of focus in the global policy arena.

Under the business as usual scenario, as the global population continues to rise, so demand for food will increase (Chapter 3). As economic development progresses, so the consumption of high P foods will increase (Chapter 8). Both of these drivers will tend to increase P losses from the food system (Chapter 5). As well as representing an economic loss to farmers (Chapters 4 and 9). Phosphorus losses from the food system will further degrade the ecosystems into which it flows, contributing to biodiversity loss and reducing ecosystem capacity to deliver services essential for sustaining life on earth (Chapter 5). Unless action is taken global P demand will remain buried within national economic development plans and undetected across the global sustainability policy arena (Chapters 2 to 9).

A more sustainable P future is essential for global food and water security. It promises economic benefits through improved natural capital (Dasgupta, 2021). The investment needed to deliver on emissions reductions, for example, using nature-based solutions, can deliver co-benefits for other pollutants. As identified by the UNEP International Resource Panel coordinated measures to reduce land based pollution will benefit coastal and transboundary ecosystems (IRP, 2021). The opportunity for improved circularity in the global P cycle is clear; currently less than about 50% of P residues are recycled back to the global food system (Figure 1.5). This represents an untapped resource in countries whose food security is exposed to the risks of high reliance on imported P fertilisers.

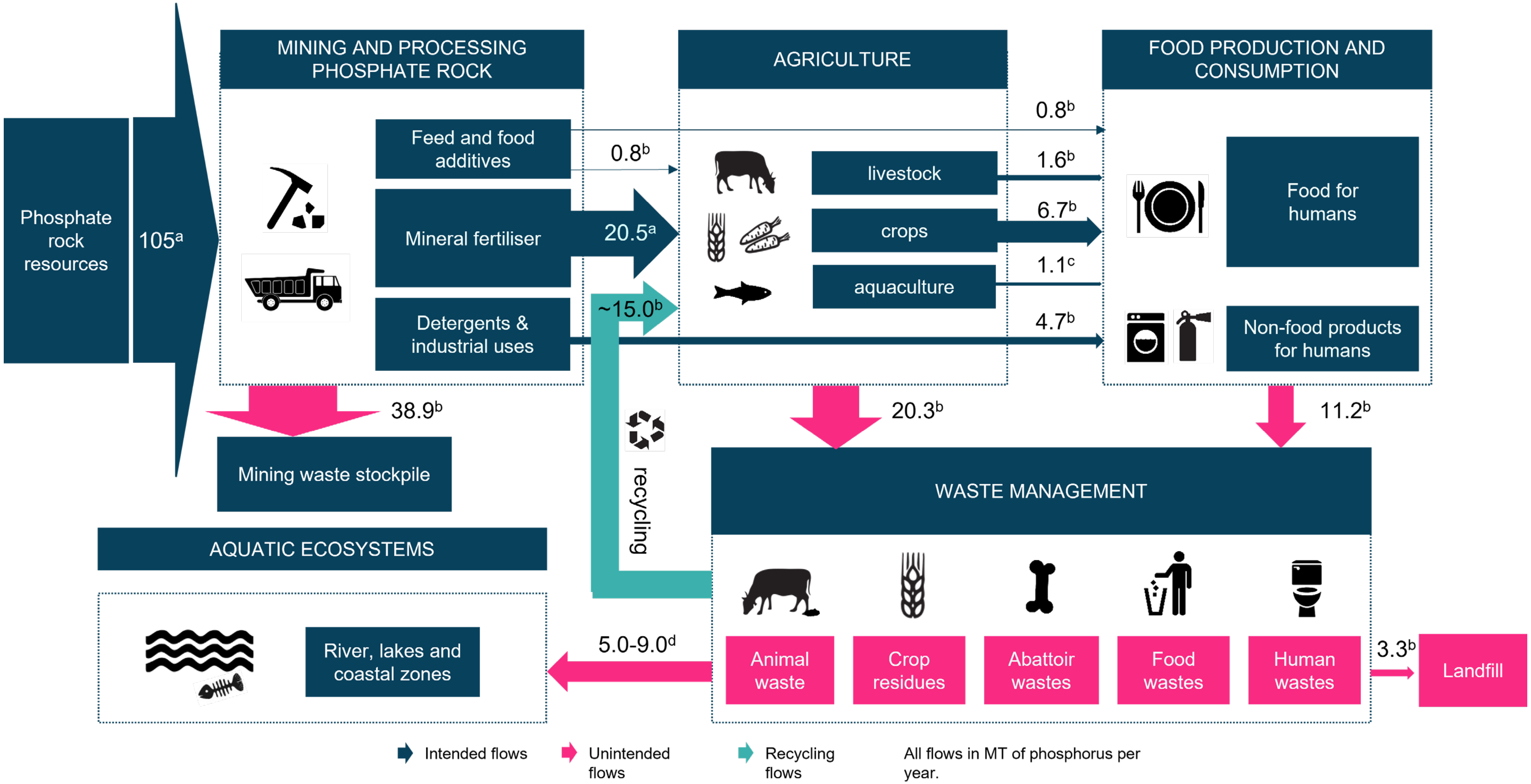

Figure 1.5 Phosphorus (Mt of P year^{-1}) flows between the key parts of the global P cycle modified from Brownlie et al. (2021). The width of arrows represents the magnitude of flows based on the current literature ([a]Jasinski, 2021; [b]Chen and Graedel, 2016; [c]Huang et al., 2020 [d]Beusen et al., 2016).

It is being increasingly acknowledged the world cannot afford to continue on its current path for phosphorus (Focus Box 1.1). In 2019 over 500 scientists signed 'The Helsinki Declaration' calling for transformation across food, agriculture, waste and other sectors to deliver much needed improvements to global phosphorus sustainability (see www.opfglobal.com to read the declaration).

This report considers the evidence required to underpin a more sustainable P future. It reflects on the need for improvements across the entire P value chain, from mine to fork, and from field to freshwater and coastal ecosystems. It considers the roles and opportunities for stakeholders in delivering such gains and the need to deliver a long-term process to track progress. Finally, an analysis of the existing policy arena is provided alongside a proposal for improved international coordination for P sustainability, identifying country-level opportunities to align P with actions on other nutrients. In doing so, a clear message emerges – that the green shoots of change are upon us. Innovation and the application of trans-disciplinary thinking are already leading to pockets of more sustainable P use, as is evident in the many pioneering case studies presented across all of the OPF Pillars.

The challenge now lies in connecting opportunities into a coherent package of measures designed to address the ultimate societal goal: to deliver global food security for a growing population, whilst reversing and preventing the destruction of the natural environment. If society is to meet this goal, then sustainable P use must be at the heart of the solution.

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02

Phosphorus reserves, resources and uses

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Left: The Bou Craa phosphate mine, taken from the International Space Station. Bou Craa is one of the largest phosphate mines in the world, and one of the few human patterns visible from space in the western extremity of the Sahara Desert. Photograph courtesy of the Earth Science and Remote Sensing Unit, NASA Johnson Space Center, www.eol.jsc.nasa.gov.

Five countries hold 85% of the planet's phosphate rock reserves. High dependency on imported phosphate rock and/or mineral phosphorus fertiliser can contribute to national food system vulnerability. Geological depletion of phosphate rock is not an immediate threat, however geopolitical, institutional, economic, and managerial factors may impact phosphorus access. Improving the efficient use of phosphorus in agriculture and shifting reliance away from mined phosphorus sources by increasing phosphorus recycling may offer the greatest protection against potential phosphorus supply risks.

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Challenge 2.1: Few nations have phosphate rock reserves

Five countries hold around 85% of known phosphate rock reserves, with 70% found in Morocco and Western Sahara alone. Most countries do not have any phosphate rock reserves and are reliant on imports to supply their phosphorus demands to maintain food security. China, Morocco and Western Sahara, the USA and Russia currently produce around 80% of the planet's phosphate rock supply.

Challenge 2.2: Phosphate rock can contain contaminants harmful to human, animal and environmental health

Different phosphate rock ores vary in their composition between phosphates, impurities and contaminants. Phosphate rock contaminants can be transferred into fertiliser products, spread on soils, and end up in food. Cadmium is of particular concern as it can pose a risk to human, animal and environmental health when above threshold levels. The by-products of phosphate rock processing also include ~200 Mt $year^{-1}$ of phosphogypsum, which can contain hazardous contaminants. Concerns have been raised that contaminant leaching from phosphogypsum stockpiles may pose a risk to the environment and the health of local communities.

Challenge 2.3: Geopolitics can impact phosphorus supply and demand, while slowing action on phosphorus sustainability

National and regional policies can have direct and indirect impacts on phosphorus access domestically or abroad. This includes taxes, tariffs, trade agreements and legislation. Political instability in countries mining phosphate rock can affect phosphate supply (e.g. Syria). Concerns over the legality and legitimacy of phosphate rock production in Western Sahara remain unresolved. Such issues also contribute to sensitivities that represent a barrier to effective dialogue and action on phosphorus.

Challenge 2.4: Phosphate rock price spikes remain an ongoing risk

In 2008, phosphate rock prices spiked by 800%, causing a subsequent increase in fertiliser prices that affected the livelihood of many of the world's poorest farmers. This price spike occurred in response to a combination of factors, including instability in energy prices, changing dynamics of supply/demand for agricultural and phosphorus products, and the influence of geopolitics on exports. The stability of phosphate rock prices remains vulnerable to such drivers.

Challenge 2.5: There is a lack of transparent, complete, and comparable phosphate rock data

Significant discrepancies in phosphate rock data are reported, making it difficult to assess accurately the risk of geographic depletion of reserves. Differing definitions for phosphate rock 'reserves' and 'resources' are a cause of discrepancies. Datasets on phosphate rock reserves and resources are commercially sensitive and are often not publicly available. Reserve estimates are dynamic and require regular updating, while conformity in data and reporting is needed. The United States Geological Survey estimates global phosphate rock reserves in 2020 at 70,000 Mt, indicating a current lifetime of >300 years, although a longer lifetime may be expected in practice due to innovation and price elasticity.

Solution 2.1: Reduce reliance on mineral phosphorus fertiliser

Replacing mineral phosphorus fertiliser with recycled phosphorus fertiliser would help to shift reliance away from mined phosphorus sources. Optimising capacity to recycle phosphorus throughout the food value chain in combination with societal change (e.g. diet change) would help to reduce phosphorus demand and losses. Enabling mainstream production of sustainable recycled phosphorus fertilisers containing low concentrations of contaminants is an essential prerequisite to upscaling operational recycling.

Solution 2.2: Establish safety levels for contaminants in fertilisers and agricultural products

Internationally agreed limits should be set for cadmium and harmful contaminants in mineral and recycled phosphorus fertilisers and food. Existing national cadmium limits require better enforcement. Optimising fertiliser use to match plant needs and practices to reduce phosphorus losses can also decrease inputs, thereby further lowering the application of fertiliser contaminants to soils, complementing the use of clean mineral and recycled phosphorus fertilisers.

Solution 2.3: Promote models of governance aimed at ensuring phosphorus security

Ensuring phosphorus security which supports all farmers to access sufficient phosphorus to grow crops, is a global responsibility and requires international cooperation. Balanced stakeholder participation in negotiations is necessary to ensure phosphate security and avoid domination of regulatory agencies by industries or private interests. An internationally agreed framework promoting sustainable phosphate rock mining and trading is currently missing and urgently needed.

Solution 2.4: Improve stakeholder capacity to deal with phosphate rock price volatility

Stakeholders need to plan for uncertainty by increasing adaptive capacity. Building national capacity to close the phosphorus loop in food production systems and shifting reliance from mined phosphorus to recycled phosphorus will help protect against phosphorus supply risk. Governments need to recognise phosphorus supply risks through appropriate policy and regulation.

Solution 2.5: Improve transparency and the independent assessment of phosphate rock data

There is a need for transparency and free access to accurate, current data on global reserves and resources of phosphate rock. An independent, international body is needed to assess data regularly and to disseminate findings through appropriate mechanisms, institutions and outreach programmes.

2.1 Introduction

For millions of years, the total amount of 'mobile' phosphorus (P) supporting the world's ecosystems remained largely unchanged. This 'mobile' P flowed between the various compartments of soil, plant, animal, wastes, waters, and sediments. Around 8000 years ago when farmers discovered that applying animal manures to croplands improved their harvests, humans started manipulating this system (Bogaard et al., 2013). In the 19th century, in addition to manures, P fertilisation of crops was commonly achieved by applying bone meal from slaughtered animals to soils (Plotegher and Ribeiro, 2016). However, as food demands increased, the low solubility of bone meal was not sufficient to provide an adequate supply of P to crops. In attempts to increase crop production, countries that could afford it replenished the P in their soils with P-rich guano (seabird and bat excrement accumulated over several millennia). A thriving industry quickly sprung up to export guano from Peru to Europe, complete with new infrastructure, overnight millionaires, and widespread worker exploitation (Melillo, 2012; Schnug et al., 2018). However, the guano deposits were quickly exhausted, and by the 1870s the guano industry had all but collapsed (Melillo, 2012; Schnug et al., 2018).

In parallel, by 1840, the agricultural scientist, Justus von Liebig had discovered the solubilisation of bone meal could be assisted through treatment with sulfuric acid (Ivell, 2012). Acid-treated bone meal released P more readily to soils and hence increased P uptake by crops (Plotegher and Ribeiro, 2016). In the same year, Liebig successfully applied the same chemical treatment to phosphate rock (PR) (Ivell, 2012). In the years that followed, John Bennet Lawes developed this process at an industrial scale, and renamed the product 'superphosphate fertiliser'; this was to be the first P fertiliser commercially produced worldwide (Ashley et al., 2011; Ivell, 2012). Since that time, with rapidly increasing production since World War II, PR has been the main source of P used by society, predominantly to provide fertilisers (Cordell et al., 2009; Ashley et al., 2011) (Figure 2.1). Over the past century, anthropogenic use of P, predominantly in agriculture, has increased by a factor of ~18 (from ~2 Mt year^{-1} in 1910 to a peak use of ~36 Mt year^{-1} in 2016) (Figure 2.1).

Globally, around 85% of phosphates produced for market are processed to make mineral P fertilisers, and 10% are used to make animal feed supplements. The remaining 5%, equivalent to ~2.5 Mt phosphorus pentoxide (P_2O_5) annually[i], is used across a range of chemical industries (de Boer et al., 2019). Of this ~2.5 Mt, around 38% is used for detergents and cleaning products, 25% for food and drink additives, 10% in metal production, 10% for water treatment to reduce dissolution of lead water pipes, 3% for specialised fertilisers (e.g. for use in aquaculture and aquaponics), and 3% for toothpaste, with the remaining 14% used for various purposes such as fuel cell electrolytes and medicines (Gantner et al., 2014; numbers rounded to nearest 1%) (Figure 2.2).

[i] Based on global consumption of ~50 Mt P_2O_5 as estimated for 2019 in Jasinski, (2020).

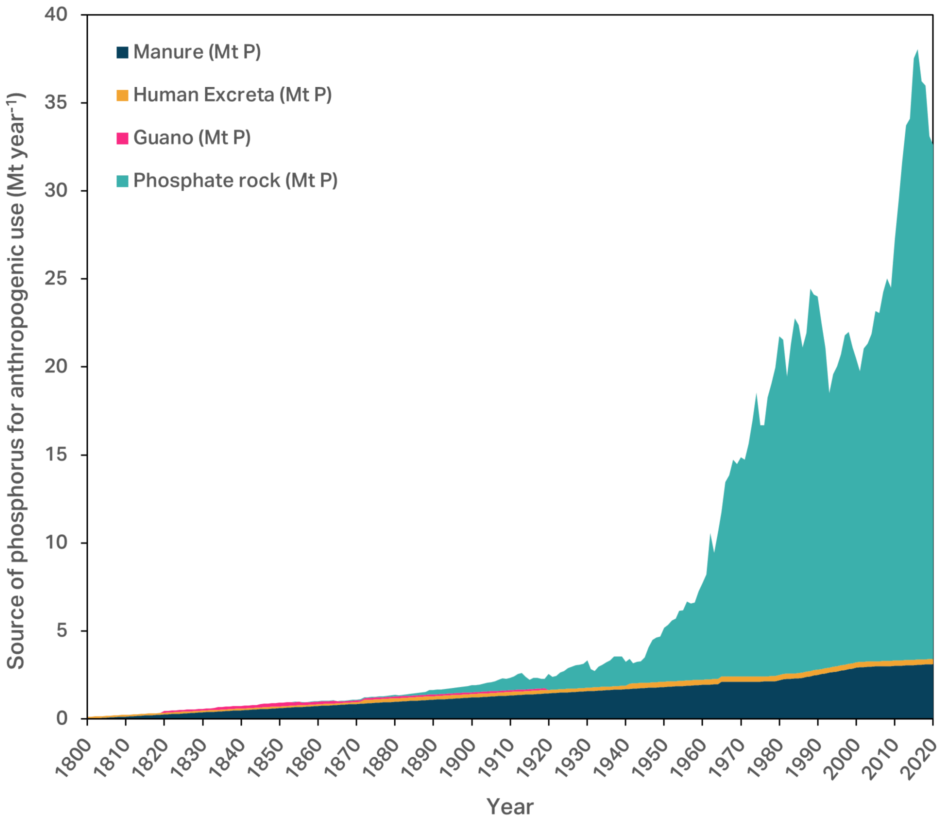

Figure 2.1 Sources of phosphorus (P) for anthropogenic use 1800–2020, including manure, human excreta, guano and P mined from phosphate rock (it is estimated ~85% of the phosphates produced from PR are used to produce mineral P fertilisers). The reliability of data sources varies; data points for human excreta, guano and manure should be interpreted as indicative rather than precise. Graph modified from Cordell et al., (2009) based on data from Smil, (2000); Cordell et al., (2009); U.S. Geological Survey, (2014); Chen and Graedel, (2016); Jasinski, (2021).

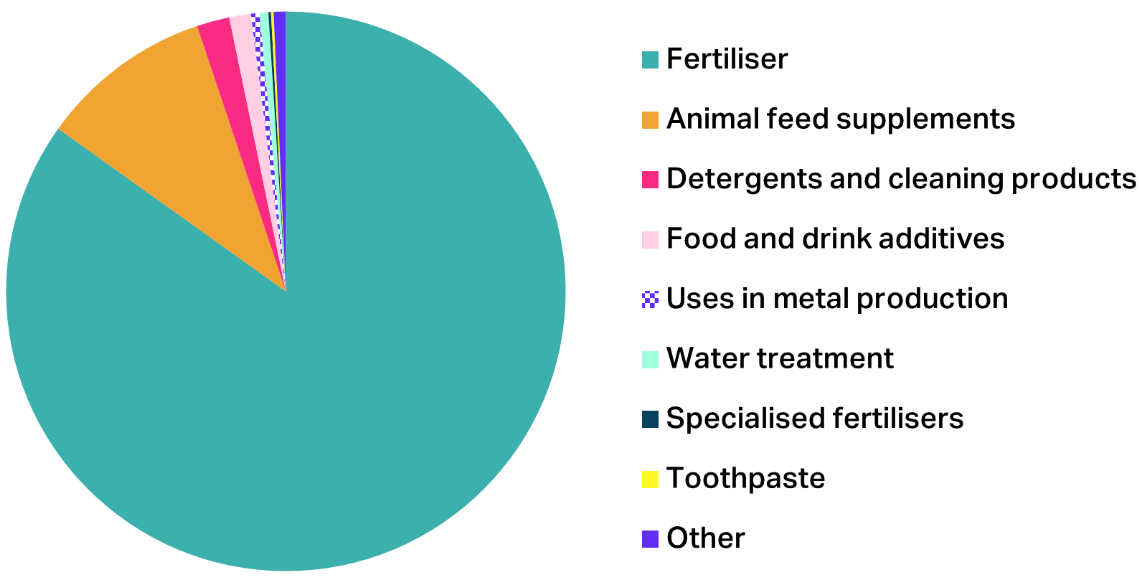

Figure 2.2 Global estimated uses of phosphorus mined from phosphate rock in 2014. 'Water treatment' refers to the addition of phosphate into potable water to reduce lead solvency from lead pipes, 'specialised fertilisers' refer to niche use fertilisers (e.g. for use in aquaculture and aquaponics). Based on data from Gantner et al. (2014) and de Boer et al. (2019).

2.2 Phosphorus sources and processing

2.2.1 Types of phosphate rock

There are three major types of PR: sedimentary, igneous and metamorphic. Each rock type has a different chemical composition, P content and range of impurities, and therefore physical characteristics (Table 2.1). Phosphate rock reserves are mined mostly at the surface, using bucket-wheel and dragline excavators and power shovels and earthmovers (Figure 2.3). The phosphate rock type has a significant impact on how easy it is to mine, the level of processing needed, and the energy requirements to complete mining and initial processing stages.

2.2.2 Processing phosphate rock and applications

Marketed PR is enriched to at least 28%, and often more than 30%, in P_2O_5. The process used to enrich PR is termed 'beneficiation' and refers to an initial processing stage, often consisting of grinding the rock, followed by flotation to separate non-P bearing minerals based on their densities and hydrophobic and hydrophilic properties.

The phosphates in untreated PR are not very water-soluble, and therefore not readily available to plants for uptake, though some plant types such as certain legumes and macadamia can mobilise the P from PR (Lyu et al., 2016; Zhao et al., 2019). Igneous deposits can provide PR with 35% to 40% P_2O_5 content after beneficiation.

Figure 2.3 Phosphate mining by the state company Société Nouvelle des Phosphates du Togo, in Togo. Photograph courtesy of Alexandra Pugachevsky.

Table 2.1 Description of the three main phosphate rock (PR) types mined for phosphorus (P), providing details on phosphate (PO_4^{3-}) concentrations (as a percentage of rock weight), mineral composition, and the location of key reserves, modified from de Boer et al. (2019) and van Kauwenbergh (2010).

Rock type	Description	Phosphate concentrations	Chemical composition	Geographic location of main global reserves
Sedimentary Rock	Sedimentary marine 'phosphorite' is the most common PR produced from accumulated fossilised shells and aquatic animals. More than 80% of mined phosphates originate from sedimentary rocks.	Phosphate concentrations in phosphatic shales and limestones range from 7.8% to 19.5% phosphate, whilst phosphorite contains above 19.5% phosphate.	Sedimentary rock contains the mineral apatite, which is calcium phosphate combined with either a hydroxide, fluoride, or chloride ion: $Ca5(PO_4)3(OH)$ or (F) or (Cl). Fluorapatite is the most common form. Whilst apatite is sparingly soluble in water, the different compositions influence its solubility.	China, Middle East, Northern Africa, USA
Igneous	Igneous deposits are associated with carbonatites and silica-deficient intrusions, formed through the crystallisation of cooling lava or magma.	Phosphate concentrations in igneous rocks are often lower than in sedimentary rocks, ranging from 0.005% to 0.4%, but higher values (i.e. up to 15% phosphate) can be found in strongly alkaline, low-silica igneous rocks.	Igneous rock mainly consists of carbonatite minerals consisting of more than 50% carbonate minerals and alkalic intrusions (e.g. diopside; $MgCaSi_2O_6$). The minerals calcite and dolomite can also be present in igneous rock.	Brazil, Canada, Finland, Russia, South Africa, Zimbabwe
Metamorphic	Both igneous and sedimentary rocks that have been subject to high temperature and pressure may form metamorphic phosphate rock.	Metamorphic rock often contains 0.01% to 1.3% phosphate.	Metamorphic rocks are created in high pressure and temperature conditions resulting in a less porous texture, more interlocked crystals, and higher induration compared to igneous and sedimentary rock. Consequently, the exploitation of metamorphic rock deposits is not currently economically viable.	China, India

The direct application of this 'granulated PR' with organic material can then be used as an alternative to mineral fertiliser, but this requires acidic soils and is not common practice (Sanchez, 2002; Chianu et al., 2012).

In most cases, the P in PR is processed into several intermediary products, including phosphoric acid and superphosphates, with less than 3% processed into (elemental) white phosphorus. Together these compounds provide the basis of industrial P chemistry (Figure 2.4).

2.2.3 Superphosphate

Superphosphate (SSP), double superphosphate (DSP) and triple superphosphate (TSP) is the term used in the P fertiliser industry for fertilisers containing monocalcium phosphate, also called calcium dihydrogen phosphate ($Ca(H_2PO_4)2.H_2O$). The key differences between SSP, DSP and TSP is their method of production and the P content of the final products. Single superphosphate is produced via the reaction between phosphate rock and sulfuric acid (H_2SO_4), whilst DSP and TSP are produced via the reaction between phosphate rock and phosphoric acid (H_3PO_4). Single, double and triple superphosphate have a P_2O_5 concentration of 7-9%, 32-36% and 44-46%, respectively.

Production of SSP is a basic technique that has changed very little since its commercialisation in the 1800s (Ivell, 2012; IPNI, 2014a). Ground PR is treated with sulfuric acid to form a semi-solid, which is allowed to cool for several hours. This plastic-like substance is then cured for several weeks. The hardened material is then milled and screened to the required particle or granule size (IPNI, 2014a). Non-granular DSP and TSP are made by reacting ground PR with liquid phosphoric acid, commonly within a cone-type mixer (IPNI, 2014b). Granular DSP and TSP are made similarly, but the resulting slurry is sprayed as a coating onto small particles to build granules of the desired size. The products from both production methods then cure for several weeks as the chemical

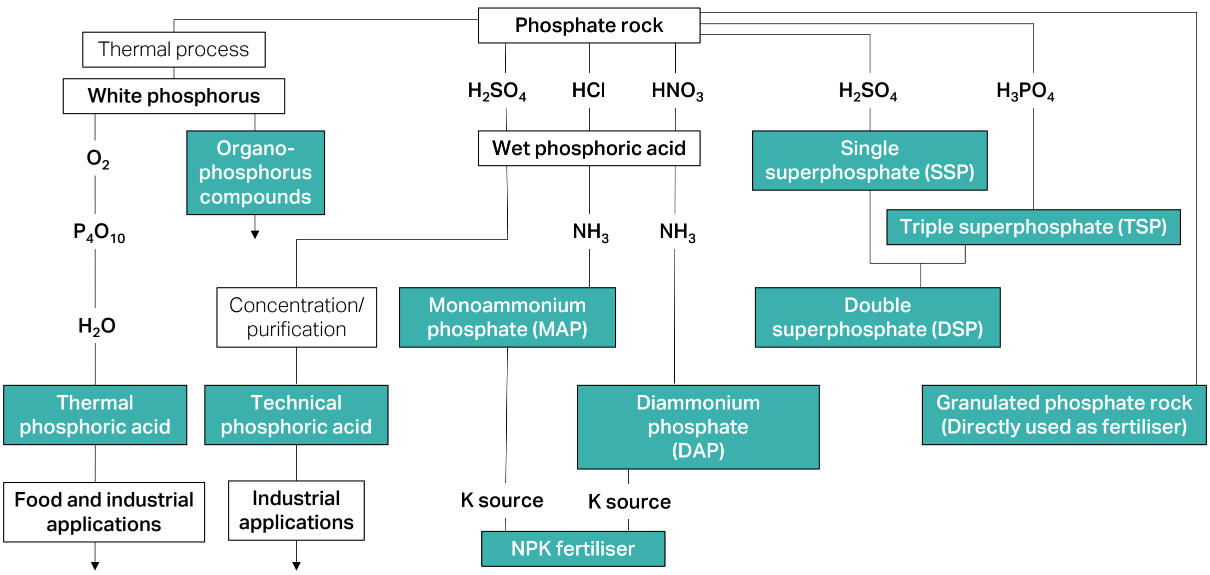

Figure 2.4 Typical processing pathways of phosphate rock to common P containing products, showing key reagents involved in the different stages. Modified from de Boer et al. (2019).

reactions are slowly completed. The chemistry and processes of these reactions vary somewhat depending on the properties of the phosphate rock.

The main fertilisers used in agriculture are monoammonium phosphate (MAP), diammonium phosphate (DAP), SSP, DSP and TSP. The ratio of nutrients differs among fertiliser types (Table 2.2).

Table 2.2 The ratio of nutrients nitrogen (N), phosphorus (P) and potassium (K) in a range of P fertilisers.

Fertiliser	Ratio of N:P:K
Monoammonium phosphate (MAP)	10:50:0
Diammonium phosphate (DAP)	18:46:0
Single superphosphate (SSP)	0:18:0
Single superphosphate (SSP)	0:36:0
Triple superphosphate (TSP)	0:46:0

2.2.4 Phosphoric acid

Two main commercial methods are used to produce phosphoric acid from PR, known as the wet process and the thermal process.

The wet process consists of three key stages: (1) digestion, (2) filtration and (3) concentration. In the first step, phosphate-containing apatite minerals (e.g. calcium hydroxyapatite or fluoroapatite) are treated in a reactor with a mineral acid (usually sulfuric acid). Nitric and hydrochloric acids (HNO_3, HCl) can also be used but are more expensive. This produces phosphoric acid and the by-product phosphogypsum (a thick slurry of solid particles containing gypsum ($CaSO_4 \cdot (nH_2O)$, where n=0, ½, or 2), unreacted residual PR and impurities. In the second stage, phosphogypsum is removed using a partial vacuum, leaving 'wet' phosphoric acid (23-33% P_2O_5). In the third step, the phosphoric acid is concentrated by reducing the liquid content, via evaporation with submerged combustion burners and vacuum circulation evaporators. 'Wet' phosphoric acid must be concentrated to 40-55% P_2O_5 for fertiliser/merchant grade, 50%-61.6% P_2O_5 for technical grade phosphoric acid, and above 61% P_2O_5 for semiconductor grade (de Boer et al., 2019).

Around 95% of the phosphoric acid produced globally is made using the wet process. An issue associated with the wet process is the production of large quantities of phosphogypsum, which can contain contaminants, depending on PR type (Tayibi et al., 2009). Impurities in PR can be carried into phosphoric acid products and/or phosphogypsum. A fourth stage can be added to remove impurities, but can represent a major challenge (Syers, 2001). The concentration and type of contaminants present are dependent on the PR origin as well as the digestion method. For example, phosphoric acid produced from igneous PR typically contains 5-10 mg l^{-1} of uranium, whilst 120-160 mg l^{-1} of uranium can be found in phosphoric acid produced from sedimentary PR (Cioroianu et al., 2001). The risk posed by impurities in fertilisers is discussed in more detail later in this chapter.

The thermal process is a less common method for producing phosphoric acid. Whilst methods differ between production plants, the thermal process can be described by three major steps: (1) combustion, (2) hydration, and (3) demisting (Speight, 2017). In the thermal process, the raw materials to produce phosphoric acid are elemental phosphorus, air and water. In the combustion step, the liquid elemental P is oxidised in a combustion chamber at temperatures of 1650–2760°C to form phosphorus pentoxide. The P_2O_5 is then hydrated with dilute phosphoric acid or water to produce a strong phosphoric acid liquid. The last stage is demisting, which removes the phosphoric acid mist from the combustion gas stream, before releasing it to the atmosphere, usually using high-pressure-drop demisters (Speight, 2017).

Because the thermal process is energy-intensive, purified wet-process phosphoric acid has replaced thermal phosphoric acid in many applications. However, phosphoric acid produced via the thermal process is of a much higher purity, which is often required to produce some organo-P compounds (often used in pesticides), such as phosphorus trichloride (PCl_3), and several other high-grade chemicals, pharmaceuticals, detergents, food products and beverages. The compounds that are produced using phosphoric acid are diverse, as summarised in Figure 2.5.

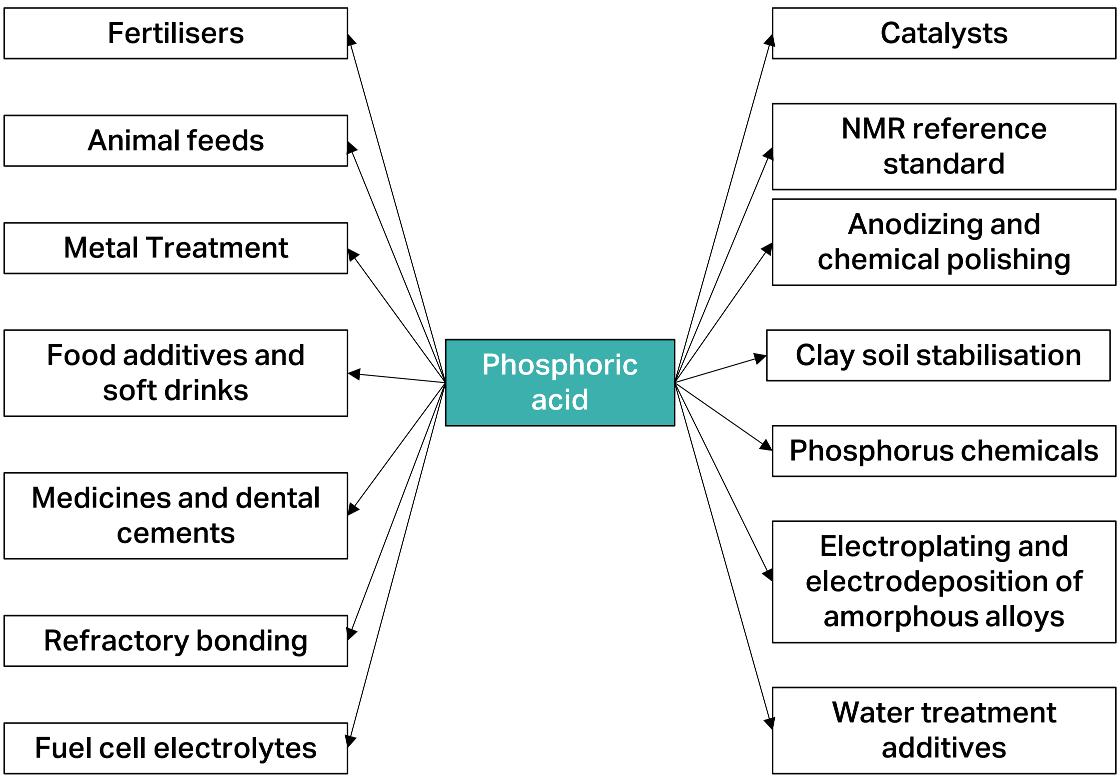

Figure 2.5 Common uses of phosphoric acid (H3PO4). NMR – Nuclear Magnetic Resonance Spectroscopy. Figure modified from de Boer et al. (2019).

2.2.5 White phosphorus

Several materials require elemental P in their production. Elemental P has ten allotropic forms, different structural modifications of an element, in which the atoms of the element are bonded together differently. The P allotrope 'white P' (P4) is used for 99% of elemental P demands, with ~0.85 Mt produced each year (de Boer et al., 2019).

White P is extremely reactive with oxygen, and as such must be stored underwater, with a layer of nitrogen above the surface. This aggressive reactivity has led to its limited and controversial use as a weapon (e.g. explosives and smoke grenades) (Macleod and Rogers, 2007). Most commonly, white P, which accounts for 1% of global P use, is a starting material to create other compounds, including those used in batteries, flame retardants, catalysts, anti-scale agents, plastic additives and the herbicide glyphosate (Figure 2.6). White P can also be further heated to make the much less reactive allotrope red P, which is stable in air and used as the striking contact on matchboxes.

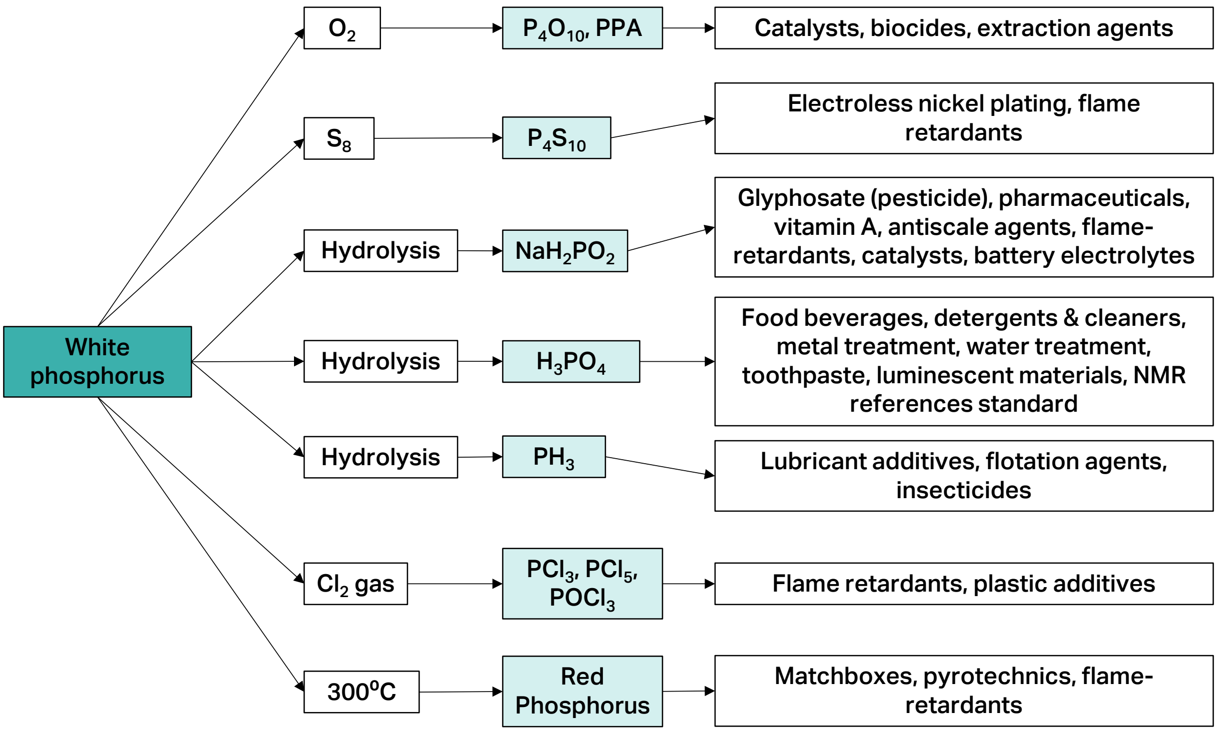

Figure 2.6 Common uses of white phosphorus, modified from de Boer et al. (2019) using data based on Gantner et al. (2014) and inputs from Willem Schipper.

2.3 Why do we need to improve the management of our phosphate rock reserves?

Demand for PR to supply fertiliser demands has increased rapidly since the 1950s, primarily due to the increase in fertilisers for agricultural production aiming to sustain a growing human population with an increasing demand for animal products (Metson et al., 2014), non-food products (Hamilton et al., 2018) and biofuel production (Jarvie et al., 2015) (Figure 2.7). A drop in production was observed between 2016 and 2020 but is still higher than at any point pre-2016 (Figure 2.7).

Besides the demand for fertiliser, there is also a steady increase in P demand for animal feed supplements, fizzy drinks, and flame-retardants (de Boer et al., 2019). Since the mid-1980s, an increasing number of countries have made reductions and/or banned the use of phosphates in domestic laundry and dishwasher detergents (but in some cases not industrial detergents) to reduce the amount of P entering waterbodies (van Drecht et al., 2009; van Puijenbroek et al., 2019). This has caused a drop in P demand for detergents over the last 3-4 decades, whilst demand in all other sectors has increased and is expected to continue to increase (de Boer et al., 2019). Drawing on data from Xu et al. (2020), Spears et al. (2022) estimate that the projected global demand for P to make $LiFePO_4$ batteries for electric cars (assuming a 60% $LiFePO_4$ market share), could increase to >3 Mt P per year by 2050.

Natural losses of P from soils to air and waters have been estimated at about 10 Mt year^{-1} (Smil, 2000). In contrast, in recent times (2000–2020), intensified erosion introduces an estimated additional 30 Mt year^{-1} P into the

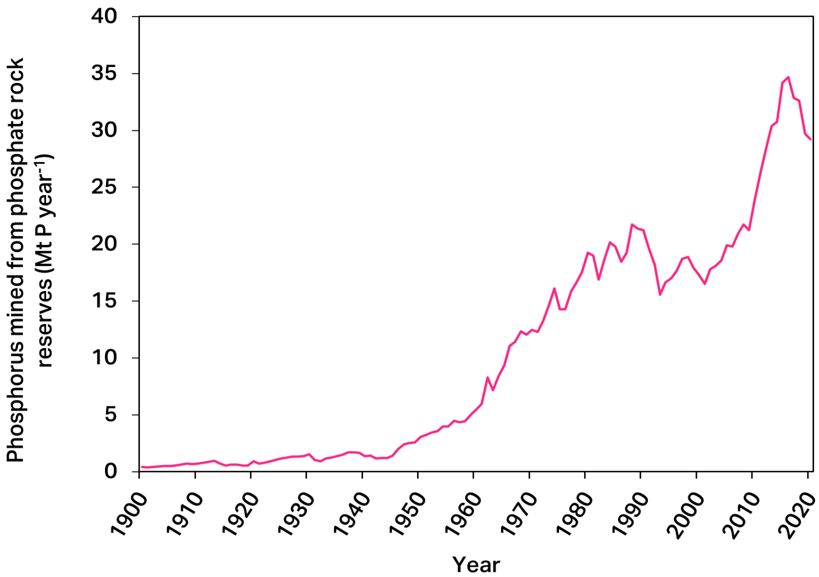

Figure 2.7 Annual mass of phosphorus mined from phosphate rock reserves globally between 1900 and 2020. Based on world production of phosphate rock (Mt) data as compiled by the U.S. Geological Survey (2014) and Jasinski (2021)[i].

[i]Data are based on the assumption that 30% of the mass of phosphate rock is P_2O_5, which was calculated as a global average of P_2O_5 content of PR mined from global PR reserves in 2017, as detailed in Jasinski (2017b). The phosphorus content of P_2O_5 was calculated by multiplying the mass of P_2O_5 by 0.4364.

global environment, mainly because human actions have approximately tripled the rate at which P reaches waters (Table 2.3). We highlight that wide variation in such estimates exists in the literature (e.g. Beusen et al., 2016). Further work is required to agree on standardised model parameters to assess P flows (see Chapter 9).

Poor P management has resulted in widespread P losses to waterbodies, causing significant damage to rivers, lakes, and coastal waters (Smith and Schindler, 2009) (see Chapter 5). Reliance on mined sources of P will continue to mobilise P that would otherwise be 'locked away', into a global mineral cycle where elevated P flows are having major impacts on freshwater and marine ecosystems around the world (see Chapter 5). The human-driven release of P has been estimated to exceed the "planetary boundary" for freshwater eutrophication (i.e., beyond levels deemed safe to avoid abrupt, irreversible environmental change) by a factor of three (Carpenter and Bennett, 2011; Steffen et al., 2015) (see Chapter 5). At the same time, current global reliance on mineral P for fertilisers to produce food means access to mineral P remains a critical requirement for food security at present (see Chapter 3). Actions to reduce the fraction of available P resources that are wasted by losses to the wider environment are necessary to address this dilemma (Chapter 6 and 7), thereby moving towards a more circular economy for phosphorus.

Table 2.3 Human intensification of the global phosphorus cycle (Mt $year^{-1}$), based on estimates from Smil (2000), updated from Chen and Graedel (2016)* and Jasinski, (2021)**. Note that estimates vary in the literature.

Flux type	Natural	Preindustrial (1800)	Recent (2000–2020)
Natural fluxes (including recent intensification by human actions)			
Erosion (wind)	<2	<3	>3
Erosion (water)	<8	>12	>27
River Transport	>7	>9	>22
Biomass combustion	<0.1	<0.2	<0.3
Anthropogenic fluxes			
Crop uptake	-	1	12
Animal wastes	-	>1	16*
Human wastes	-	0.5	3
Organic recycling	-	<0.5	15*
Mineral fertiliser	-	-	21**

2.4 Global phosphorus reserves and resources

Whilst estimates vary, in 2020, 29.2 Mt of P was mined from global PR reserves estimated to contain 9295 Mt of P (Jasinski, 2021)[i]. For the purpose of these estimates, PR 'reserves' are defined as PR deposits from which PR can be economically produced at the time of the determination using existing technology. This definition can be contrasted with P 'resources', which are defined as PR of any grade, including deposits that cannot be currently mined without significant economic and/or environmental cost (van Kauwenbergh, 2010).

Based on rates of PR production for 2020, the 'lifetime' of existing reserves is 318 years (Jasinski, 2021). However, this estimate fluctuates in response to changes in reported estimates (van Kauwenbergh et al., 2013; Blackwell et al., 2019). For example, annual PR production and PR reserves data for 2010 indicated PR reserves would last for 87 years. One year later this estimate rose to 325 years, largely due to an increase in reserve estimates in Morocco (discussed later in this chapter) (Jasinski, 2009) (Figure 2.8).

Estimated PR production can be compared with global consumption of phosphoric acid, fertilisers, and other uses, which in 2020 contained 21 Mt of P, 68% of the amount of PR mined (i.e. 47 Mt of P_2O_5 (Jasinski, 2021)). The remaining 32% is estimated to be lost to the environment as part of PR processing.

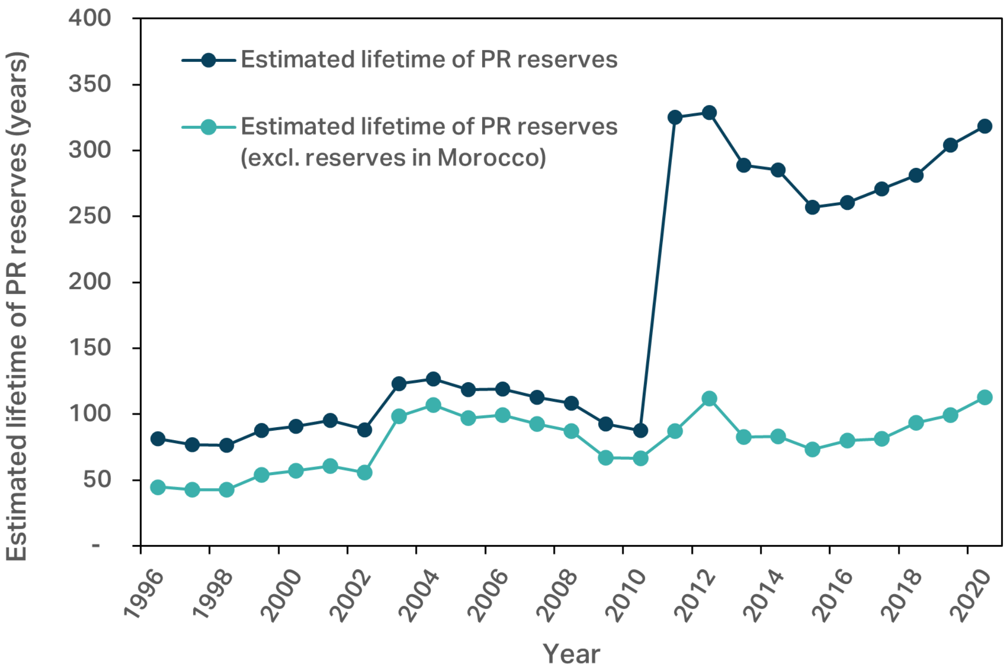

Figure 2.8 Estimated lifetime of global phosphate rock (PR) reserves (years) at current rates (2021) by year between 1996 to 2020, shown with and without Morocco. The estimated lifetime of reserves excluding Morocco is shown based on current total global production assuming that this is market driven.

[i] Calculated from Jasinski (2021) and assumption that 30% of the mass of phosphate rock (PR) is P_2O_5, as a global average, as detailed in Jasinski (2017b), i.e. in 2020, 223 Mt of PR, containing 67 Mt P_2O_5, was mined from global PR reserves of 71,000 Mt of PR, containing 21,300 Mt P_2O_5. Lifetime estimates also assume that remaining PR reserves are of homogeneous P_2O_5 content of 30% by weight. The phosphorus content of P_2O_5 is calculated by multiplying by the mass of P_2O_5 by 0.4364.

Steiner et al. (2015) showed that PR mining efficiency increased between 1983 and 2013, due to improved technologies to regain losses during the excavation and early PR processing stages. They also suggested that the average P content of PR mined globally increased from 513 Mt at 14.5% P_2O_5 in 1983, to 661 Mt at 17.5% P_2O_5 in 2013. However, PR reserves are finite, and concerns that high-quality PR reserves will diminish over time have been raised (Cordell and White, 2011; Scholz et al., 2013; Edixhoven et al., 2014).

Changes in estimated reserves have been observed for other commodities, for example, oil (Aleklett, 2012), rare earth elements, magnesium, tin and tantalum (Sutton et al., 2013; Scholz and Wellmer, 2013). However, unlike many other commodities where alternatives are available (e.g. renewable energy provides an alternative to fossil fuel combustion), nutrients are essential biological requirements; there are therefore no alternatives. Furthermore, nitrogen as a component of the air, is not constrained geographically.

Total global PR resources are estimated at 40,000 Mt of P (or 300,000 Mt of PR). As reserves are depleted and technological innovation occurs, some of these resources would be expected to be made available to increase the number of minable reserves. Based on current mining rates of 30 Mt P year^{-1}, the lifetime of estimated PR resources is 1330 years. There is therefore no risk of exhausting global P supplies within the next 100-300 years (Blackwell et al., 2019). However, Blackwell et al. (2019) demonstrate that, if mining production continues at the current rate (as of 2018 and based on current reserve estimates), domestic supplies in the three countries with the largest populations, China, India, and the USA, will be depleted within 40 years. A similar calculation for 2020 estimates (of PR reserves and production) shows that, within the next 46 years, three of the four countries that produce the highest quantities of PR globally (i.e. China, the USA and Russia), will have exhausted their PR reserves (Table 2.4). While some of these time horizons may appear long, it is evident that such a linear economy of mining followed by waste of lost P resources is ultimately unsustainable, with a major challenge to move toward a more circular system where recovery and reuse are central.

The calculated lifetimes of reserves summarised in Table 2.4 are unlikely to be accurate in practice. As demonstrated above, reserve estimates fluctuate, new reserves may be found, and market forces and technological advances may allow further resources to be economically mined (i.e. converting resources to reserves). However, Table 2.4 highlights that imminent, fundamental changes in global P trade, use and recycling efforts will be necessary. Whilst most now acknowledge that geological depletion of PR is not an immediate threat, factors that may impact P access go beyond physical reserves and include geopolitical, institutional, economic and managerial factors (Cordell and White, 2011).

In the following section, the challenges and solutions are discussed for managing our PR reserves in ways that are more sustainable, equitable and safe for human and environmental health. We highlight potential P access issues and risks to supply and suggest key actions that will help to improve 'Phosphorus Security'. As defined by Cordell, (2010), 'Phosphorus Security' allows the world's farmers to access sufficient P in the short and long term to grow enough food to feed a growing world population, while ensuring farmer livelihoods and minimising detrimental environmental and social impacts.

Table 2.4 Estimated phosphate rock (PR) reserves in 2020, PR production in 2019, with a lifetime of reserves calculated at 2020 production rates. The countries selected include the ten countries that produced the highest quantities of PR in 2019 and India. Based on data from Jasinski (2021).

Country/region	PR Reserves (Mt)	Annual PR Production (Mt)	Lifetime of reserves at 2019 production rates
China	3200	90.0	36
USA	1000	24.0	42
Morocco and Western Sahara	50000	37.0	1351
Russia	600	13.0	46
Jordan	800	9.2	87
Tunisia	100	4.0	25
Brazil	1600	5.5	291
Egypt	2800	5.0	560
Israel	57	2.8	20
Australia	1100	2.7	407
India	46	1.5	31
World	**71,000**	**223**	**318**

2.5 Challenges

Challenge 2.1: Few nations have phosphate rock reserves

Five countries hold around 85% of known phosphate rock reserves, with 70% found in Morocco and Western Sahara alone. Most countries do not have any phosphate rock reserves and are reliant on imports to supply their phosphorus demands to maintain food security. China, Morocco and Western Sahara, the USA and Russia currently produce around 80% of the planet's phosphate rock supply.

Phosphate rock reserves are not equally distributed in the world (Jasinski, 2021) (Figure 2.9). Around 85% of the world's known PR reserves are found in only five countries, with 70% in Morocco and Western Sahara alone (Jasinski, 2021).

Mining in China, Russia, the USA, Morocco and Western Sahara accounted for around 80% of global PR production in 2019 (Jasinski, 2021). Since 2006, the production of PR in China has increased significantly, from 30.7 Mt to a peak of 144 Mt in 2017 (Jasinski, 2009, 2019) (Figure 2.10).

Whilst China contains less than 5% of the global PR reserves, it is the largest producer (and consumer) of PR in the world, accounting for 52% of global PR production in 2019 (Jasinski, 2021) (Figure 2.11). However, with ongoing investments in PR mining in Morocco it is anticipated that Morocco will become the largest producer in the coming years (Rosemarin and Ekane, 2016).

In 2019, the global phosphate fertilisers market was worth US$66 billion, with growth projected at a compound annual rate of 7% to reach US$84 billion by 2023 (The Business Research Company, 2020). The global market is dominated by a few key companies (Rosemarin and Ekane, 2016). In 2021, major players in the phosphate fertilisers market included Agrium Inc. (Canada), Coromandel International Ltd. (India), EuroChem Group A (EU), Guizhou (China), Israel Chemicals Limited (Israel), Ma'aden (Saudi Arabia), The Mosaic Company (USA), Nutrien Ltd. (Canada), OCP S.A (Morocco), PhosAgro (Russia) and Yunthianhua (China).

A high dependency on imported PR and/or mineral P fertiliser can contribute to national food system vulnerability. For example, South Asia is almost completely reliant on P imports (Subba Rao et al., 2015; Jasinski, 2018). In 2015, India imported 8.27 Mt of PR, mainly from Jordan (39%), Egypt (22%) and Morocco (17%), whilst 85% of India's mineral P fertilisers came from China (Indian Bureau of Mines, 2016). Similarly, European countries rely heavily on P imports for mineral fertiliser and animal feed supplements (Ott and Rechberger, 2012; van Dijk et al., 2016). In 2010, the EU imported 7.5 Mt (de Ridder et al., 2012), whilst the remaining ~10% was produced in Finland (the only active PR mining country in Europe). Similarly, Australia is heavily reliant on imports, which have been estimated to supply 80% of its P use (Cordell et al., 2013).

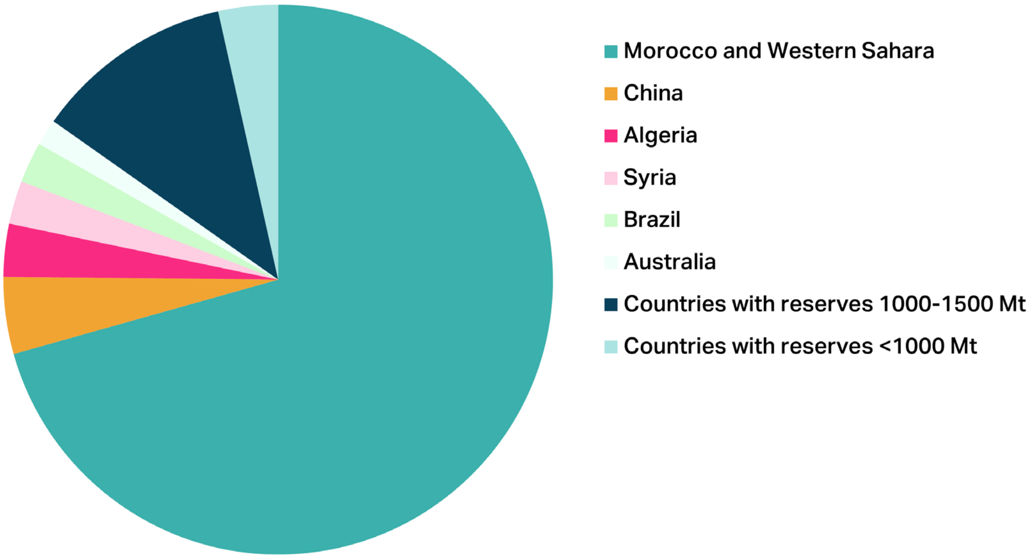

Figure 2.9 Distribution between countries of known phosphate rock (PR) reserves in 2019, equal to 69,000 Mt. Countries with reserves 1000-1500 Mt are Finland, Jordan, and the USA. Countries with <1000 Mt include Russia, Peru, India, Senegal, Kazakhstan, Tunisia, Uzbekistan, Vietnam, Mexico, Israel, and Togo. (Data source: (Jasinski, 2021)). Phosphate rock 'reserve' is defined here as the part of the reserve base which could be economically extracted or produced at the time of determination but need not signify that extraction facilities are in place and operative.

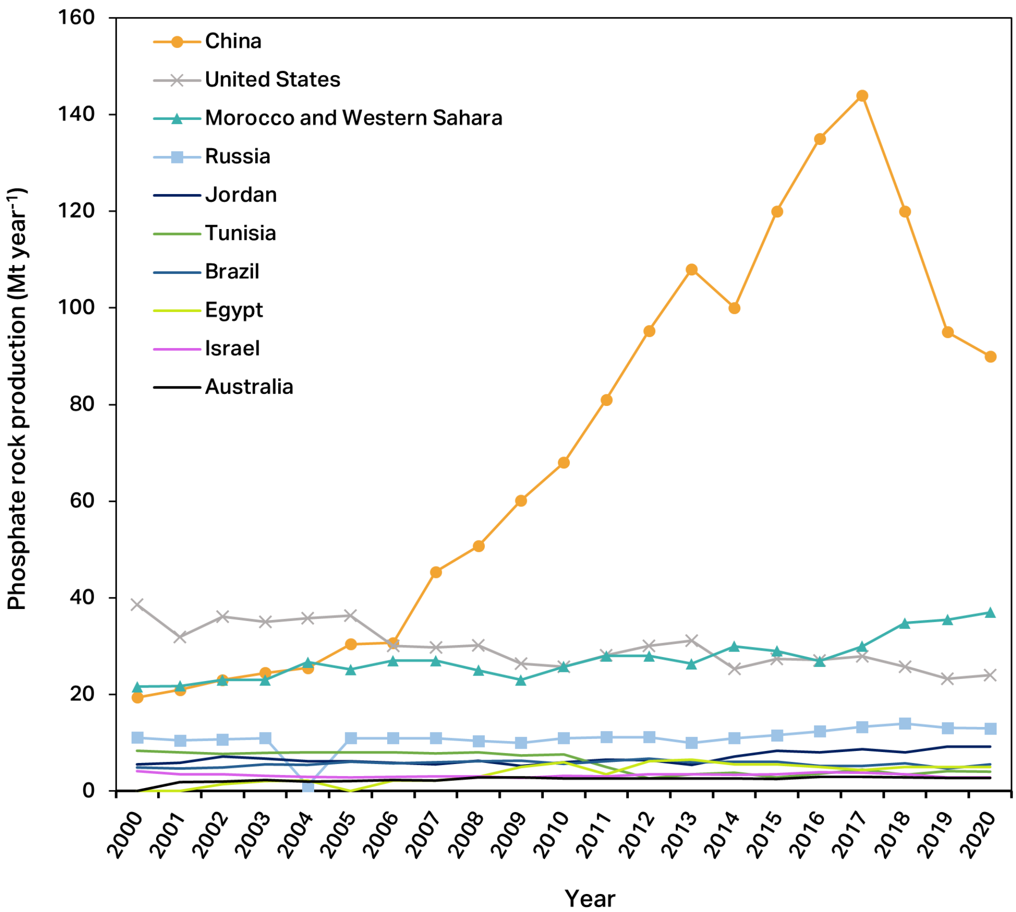

Figure 2.10 Phosphate rock (PR) production between 2000 and 2020 in the ten countries that produced the highest quantities of PR in 2020. Based on data from the annual U.S. Geological Survey reports from 2003-2021 (Jasinski, 2003 through to 2021).

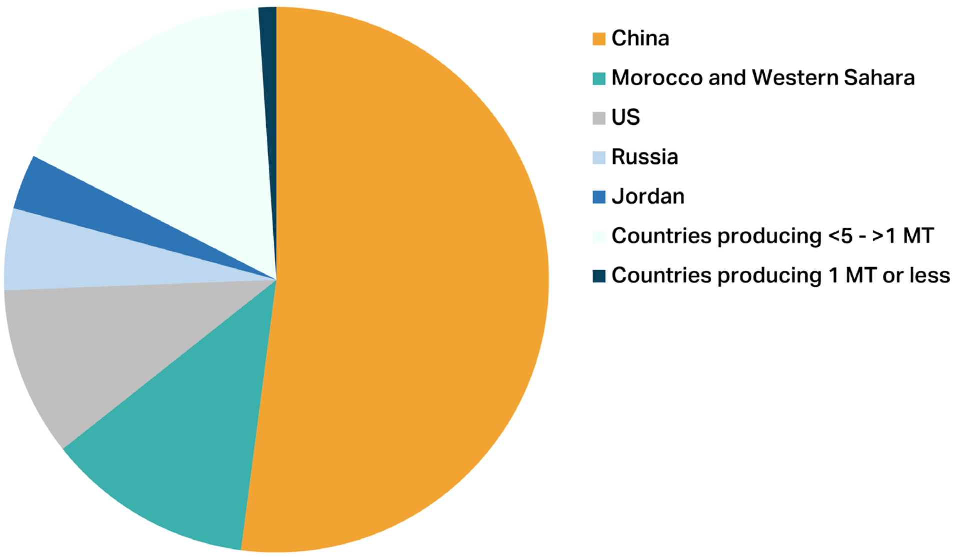

Figure 2.11 Distribution between countries of global PR production in 2019 (Data source: Jasinski, 2021).

China, the USA, Russia, Saudi Arabia and Morocco are among the few countries that contain large PR reserves and fertiliser production facilities, and hence are less reliant on imported phosphorus.

For P, the concept that 'the few' own the supply needed by 'the many' is an oversimplification. Not all countries with PR deposits mine them, while not all countries that mine PR can produce fertilisers. In addition, some countries that lack PR reserves are producers of P fertilisers (Table 2.5). For example, phosphate rock resources are located across Europe in Belgium, Germany, Spain, France, Italy, Greece and the UK. However, under current markets, these resources are not economically viable to mine and/or have been closed due to environmental concerns (Notholt et al., 2005). Brazil contains 2.4% of known global PR reserves, producing 2.0% of global PR annually (FAO, 2017). However, Brazilian PR reserves are mainly poor quality igneous fluorapatites, with low P solubility and high processing costs.

Although Brazil has been expanding its PR production, it is insufficient to supply domestic requirements, and thus imported P supplies about 70% of Brazil's mineral P fertiliser demands (Withers et al., 2018b).

Some countries may contain PR reserves but lack the infrastructure to produce fertilisers. For example, Sub-Saharan African countries (SSA) hold an estimated 2% of global PR reserves (Jasinski, 2021), but manufacture little fertiliser (Chianu et al., 2012). Phosphate rock is instead exported from SSA, often to EU countries that do not have PR reserves but do have fertiliser production capacity. In some cases, fertiliser produced in the EU is re-imported back into SSA (de Ridder et al., 2012), where it is sold at prices reported to be too expensive for most farmers to access (Chianu et al., 2012). By contrast, despite having no significant PR reserves, Japan produces ~36% of the fertiliser it uses for domestic agriculture and even exports a small amount of mineral P fertiliser (FAO, 2017).

Table 2.5 Phosphate Rock (PR) reserves and PR mine production for 2018 (Jasinski, 2021) and the estimated P imported and produced and contained in all P products in 2016 (FAO, 2017) for different regions/countries. Percentage of P supplied by imports (I), or produced (Pr) within the country, followed by the percentage of the total supply that was used in the agricultural sector (A), used outside of the agricultural sector (O) or exported (E). FAO data assume I + Pr = A + O + E. Discrepancies in the equation balance may reflect a change in P stock within the county or the available data was collected from different years. For further details see supporting documents for data provided by FAOSTAT.

Region	PR Reserve (% of global reserves)	PR Mine production (% of global production)	Import (I) + Production (Pr) = Agricultural Use (A) + Other Uses (O) + Export (E)						
			I	Pr	I+Pr (Mt P)	A	O	E	A+O+E (Mt P)
European Union	1%	<1%	75%	25%	2.84	40%	23%	37%	2.80
Brazil	2%	2%	68%	32%	2.08	95%	4%	1%	2.28
US	1%	10%	20%	80%	4.44	51%	26%	23%	4.45
Australia	2%	1%	63%	37%	0.64	79%	0%	21%	0.64
India	<1%	1%	32%	68%	2.92	99%	0%	1%	2.94
SSA	2%	2%	36%	64%	0.81	38%	34%	28%	0.81
China	5%	52%	2%	98%	7.75	76%	0%	24%	9.10
Japan	0%	0%	64%	36%	0.24	98%	0%	2%	0.15
Syria	3%	<1%	10%	90%	0.01	34%	0%	66%	0.01
Morocco	71%	12%	0%	100%	0.85	4%	0%	96%	2.31

The European Commission has formally acknowledged that PR is a commodity with a high supply risk, adding PR to the EU 'Critical Raw Materials list' in 2014 (European Commission, 2014) and white P in 2017 (European Commission, 2017). The EU Critical Raw Materials list is composed of 30 materials (after 2020 revision) of high importance to the EU economy and with a high risk associated with their supply (European Commission, 2020). These decisions were made because:

- limited PR reserves in the EU make it highly dependent on imports,
- P demand is expected to increase (due to the growing world population), and
- there are no alternatives to P in fertilisers or animal feeds (European Commission, 2014).

Challenge 2.2: Phosphate rock can contain contaminants harmful to human, animal and environmental health

Different phosphate rock ores vary in their composition between phosphates, impurities and contaminants. Phosphate rock contaminants can be transferred into fertiliser products, spread on soils, and end up in food. Cadmium is of particular concern as it can pose a risk to human, animal and environmental health when above threshold levels. The by-products of phosphate rock processing also include ~200 Mt year^{-1} of phosphogypsum, which can contain hazardous contaminants. Concerns have been raised that contaminant leaching from phosphogypsum stockpiles may pose a risk to the environment and the health of local communities.

Contaminants of PR include non-P bearing minerals and potentially toxic elements, such as heavy metals and metalloids. Heavy metals found in PR include arsenic, cadmium, chromium, uranium, mercury, lead, iron and copper (Al-Shawi and Dahl, 1999; Corbridge, 2000). Phosphate rock can also contain the radionuclides uranium and thorium. The recovery of uranium from PR can have value for nuclear power production. The USA recovered uranium from PR processing from the 1970s to the 1990s, but the production facilities were dismantled when prices for uranium from other sources dropped (Merkel and Hoyer, 2012). Processing PR into fertiliser can increase the concentration of potentially toxic elements by 1.5 times compared to the original ore (Sattouf, 2007). Cadmium concentrations in fertilisers have been reported between 1 and >640 mg kg^{-1} (Mclaughlin et al., 1996; Ulrich et al., 2014). The concentration of contaminants depends on the origins of the PR and is also highly variable between and within the same PR reserve. Typically, concentrations of heavy metals and radioactive elements are much higher in sedimentary phosphate rock. However, whilst igneous rock deposits have lower contaminant concentrations, they also contain less phosphate. The PR reserves in Finland and Russia are igneous and relatively contaminant-free, whereas most PR ores in Africa, the Middle East and Israel are sedimentary and contain higher concentrations of contaminants (e.g. cadmium) that are largely exported, transferred into fertiliser products and spread on soils (Hermann et al., 2018). Russia is the only major PR producing country with mainly igneous rock reserves. Differences in the content of cadmium and phosphate in PR ores between the major PR producing countries, China, Russia, the USA and Morocco, are summarised in Table 2.6.

Cadmium, uranium, chromium and arsenic tend to bio-accumulate (Al-Shawi and Dahl, 1999; Sabiha-Javied et al., 2009; Ertani et al., 2017) and can be toxic to humans and animals if ingested above certain levels (Kratz et al., 2011). Cadmium is efficiently retained in the kidney and liver in the human body, with a very long biological half-life ranging from 10 to 30 years (EFSA, 2012). Cadmium is classified as a class 1 carcinogen (Group B1) by the International Agency of Research on Cancer and the World Health Organization (World Health Organization & International Programme on Chemical Safety, 1992; IARC, 1993). Phosphate fertilisers are the main source of uranium enrichment of topsoil globally (Schnug and Lottermoser, 2013). Although little uranium is incorporated into crops, it can bind to root vegetables, and thus enter the food chain, and also be transported by groundwater and surface waters. The intake of uranium by humans from plant products is considered negligible, with ingestion through drinking water suggested as the main exposure route (Schnug et al., 2005). Whilst it is dependent on the region, uranium in drinking water may derive from the bedrock geology or from fertiliser use (Smedley et al., 2006; Brindha et al., 2011; Smidt, 2011).

Concentrations of cadmium and other metals in soils vary and depend on natural background levels, but increasingly also on human activities, including mineral

Table 2.6 Concentrations of cadmium (Cd) and phosphate in phosphate rock (PR) ore types of the major PR producers, China, Morocco, US and Russia. Modified from de Boer et al. (2019b), with cadmium data based on Mar and Okazaki (2012) and Geissler et al. (2019).

Origin of PR reserves	PR ore type	Cadmium content (mg Cd kg $rock^{-1}$)	Phosphate content
China	Sedimentary	4	High
Morocco	Sedimentary	3-186	High
USA	Sedimentary	3-186	High
Russia	Igneous	0.1-<13	Low-moderate

fertiliser use (Römkens et al., 2018). From analysis of mineral P fertilisers used in Germany and Southern Brazil, (Smidt et al., (2011) estimated that annually, 42 and 611 t of cadmium, and 228 and 1614 t of uranium, are applied to agricultural soils in mineral P fertilisers in Germany and Brazil, respectively. This is equivalent to 2.9 g cadmium ha^{-1} $year^{-1}$ and 11.8 g uranium ha^{-1} $year^{-1}$ applied to agricultural soils in Brazil, and 1.4 g cadmium ha^{-1} $year^{-1}$ and 8.2 g uranium ha^{-1} $year^{-1}$ applied to agricultural soils in Germany (Smidt et al., 2011). With repeated application of fertilisers containing cadmium, levels of cadmium can accumulate to undesirable concentrations in agricultural topsoil, dependent on multiple factors including fertiliser application rates, crop rotation, soil properties and weather conditions (Bigalke et al., 2017). Cadmium sequestration by some crops can be high, depending on soil pH, organic matter, and leaching losses to deeper soil layers or in run-off (Smolders, 2001; Rizwan et al., 2017). FitzGerald and Roth (2015) concluded that cadmium in fertilisers used in Switzerland should be controlled to reduce human exposure as much of the population already ingests close to the tolerable limit. The European Food Safety Authority (EFSA) has shown that children and adults that consume higher amounts of vegetables grown in soils containing cadmium frequently exceed the tolerable weekly intake of 2.5 µg cadmium kg^{-1} body weight (EFSA, 2012). Although the risk of adverse effects on kidney function at an individual level at dietary exposures across Europe is very low, it was concluded that the current exposure to cadmium at the population level should be reduced (EFSA, 2012). A study by de Vries and McLaughlin (2013) suggested that cadmium inputs in fertilisers in Australia exceed the long-term critical loads in heavy-textured soils for dryland cereals, although all other systems are at low risk. de Vries and co-authors concluded that current cadmium inputs in fertilisers in the EU27 countries are nearly always below critical thresholds for toxic impacts on food quality (ETC/ULS, 2016) and soil organisms (de Vries and Römkens, 2017), but exceedance may occur locally. Similarly, Römkens et al. (2018) predicted small absolute changes at the EU level in soil cadmium concentration of less than 0.02 mg kg^{-1} after 100 years application of P fertilisers with cadmium concentrations of 20 and 60 mg cadmium kg^{-1} P_2O_5. However, regional differences were substantially larger varying from -0.15 mg kg^{-1} to 0.07 mg kg^{-1} (compared to the EU average soil cadmium content).

As mentioned earlier, phosphogypsum is a by-product of the digestion stage in PR processing. Production of 1.0 t of phosphoric acid yields around 5.0 t of phosphogypsum, equivalent to a global annual production of 100-280 Mt phosphogypsum $year^{-1}$ (Tayibi et al., 2009; Saadaoui et al., 2017). The construction industry uses gypsum ($CaSO_4$) as a component of cement amongst other uses. For example, in Russia, between 2008 and 2019, hemi-hydrate phosphogypsum was used in road building (Levin et al., 2020). However, impurities in phosphogypsum, such as thorium, radium, cadmium and uranium, can prevent such uses depending on the regulatory threshold levels for contaminants within the operating country. Hilton (2020) provides an expansive list of regulatory and commercial barriers related to phosphogypsum use, by country. 'Flotation', which is mainly used to separate P from igneous rock deposits (see Section 2.2.2 above), is increasingly used to extract P from 'low grade' sedimentary ores, that were traditionally discarded as waste, thus reducing stockpiling of phosphogypsum (Steiner et al., 2015). However, currently, around 85% of phosphogypsum produced in the wet process is disposed of without treatment (de Boer et al., 2019), and discharged into the sea and watercourses, or stored in large stockpiles. The stockpiling of large quantities of potentially hazardous phosphogypsum generates concerns over impacts to the environment and health risks for communities living close to stockpiles (Rutherford et al., 1994; Tayibi et al., 2009; Saadaoui et al., 2017; Attallah et al., 2019). The composition and characterisation of phosphogypsum depend mainly on the ore source (Hilton, 2020). Where phosphogypsum is free from or contains insignificant amounts of contaminants, it may have much fewer negative impacts on the environment.

The leaching of toxic elements and radionuclides from phosphogypsum stockpiles shows variation been multiple sites that have been assessed. For example, minimal environmental impacts were observed from stockpiles in Poland (Olszewski et al., 2016), Jordan (Al-Hwaiti, 2005) and Greece (Papageorgiou et al., 2016), whilst more significant impacts were observed from stockpiles in India (Haridasan et al., 2001) and Portugal (Corisco et al., 2017). The method used to assess the impacts of different stockpiles will influence comparability between results. However, differences can be largely attributed to the variation in impurity concentrations of the phosphogypsum stockpiled, and the local conditions (e.g. weather, soil types, hydrology, species impacted), suggesting that thorough and individual site-level assessments will always be needed to reduce risks.

Challenge 2.3: Geopolitics can impact phosphorus supply and demand while slowing action on phosphorus sustainability

National and regional policies can have direct and indirect impacts on phosphorus access domestically or abroad. This includes taxes, tariffs, trade agreements and legislation. Political instability in countries mining phosphate rock can affect phosphate supply (e.g. Syria). Concerns over the legality/legitimacy of phosphate rock production in Western Sahara remain unresolved. Such issues also contribute to sensitivities that represent a barrier to effective dialogue and action on phosphorus.

Geopolitics (i.e. the interaction between politics and international relations and dynamic geographical settings) can have significant impacts on P supply and demand. Countries with the capacity to export PR or P fertiliser can implement export restrictions in the form of export taxes, quantitative restrictions, or export bans of P products (PR and fertilisers) (Karapinar, 2011). Such export restrictions are often subject to extensive public attention and heated debate as they can impact food security (Karapinar, 2011).

For example, in 2008, China introduced an export tax of 100-135% on fertilisers to ensure that fertilisers produced in China were used domestically (Huang, 2009). This was driven by an increase in domestic fertiliser demand to match increased national agricultural production, and by concerns that easily extracted and processed Chinese PR reserves were being overexploited (van Kauwenbergh, 2010; de Ridder et al., 2012; Li et al., 2015). In relation to P specifically, this led to legal proceedings under the World Trade Organisation (WTO). China is permitted under its WTO obligations to impose an export duty of up to 20% on yellow phosphorus. In early 2009, it had imposed an additional duty of 50% on yellow P, as well as certain other measures, including minimum export prices. Because of this, and related restrictions on other raw materials, including PR (de Ridder et al., 2012), the EU, Mexico and the USA commenced legal proceedings in the WTO against China in 2009. China withdrew the special duty of 50% on yellow P on 1 July 2009 before the dispute commenced and so the WTO panel made no findings on this point, but China lost its case in relation to minimum export prices for yellow P (World Trade Organization, 2009). In the end, China succeeded on appeal on a technicality, namely that the three complaining countries had not specified the legal basis of their claim with sufficient precision (World Trade Organization, 2009). In January 2019, China dropped all taxes on all fertiliser exports, including phosphate ore and phosphoric acid.

Policies in P-importing countries can also affect supply. In late 2018, the European Parliament agreed on the revision of Fertilisers Regulation (EC) No 2003/2003, originally proposed in 2016. The revised 'Fertilising Products Regulation' (EU) 2019/1009, which repealed Regulation (EC) No 2003/2003 and harmonised the requirements for phosphate fertilisers including by setting harmonised cadmium limits (Fertilising Products Regulation, Part II), may increase the EU's dependence on countries that can provide low cadmium PR or fertilisers (discussed in more detail in Solution 2.1 below).

Instability in the Middle East and North African countries after the 'Arab Spring' has been noted to impact P trade, with issues over sovereignty, labour disputes and civil war cited as factors impacting PR production (Webeck et al., 2014; Smith, 2015). In Tunisia, annual PR production decreased from 7.6 to 2.6 Mt between 2010 and 2012 (Jasinski, 2013), related to unrest among industrial workers, due to employment disputes, involving miners, as well as rail workers, who obstructed the transport of PR from mines (Gobe, 2010; de Ridder et al., 2012; Al Jazeera, 2020). Previously, the Syrian PR industry, which held 1800 Mt in PR reserves produced 3.9 Mt of PR in 2006 (Jasinski, 2009). However, the phosphate rock mines in Iraq and Syria were closed in late 2015 because of ongoing conflicts (Jasinski, 2009). Conversely, during the same period, other countries in the Middle East increased PR production, including Egypt and Saudi Arabia, where production doubled between 2011 and 2013 (Jasinski, 2013). It could be argued that this reflects an automatic rebalancing mechanism on the supply side: when production in one area falls, it is compensated by a rise elsewhere. Disruption in Syria and Tunisia had effectively no impact on the global price of phosphate rock. However, this is expected since the highest PR production levels in the period 2000-2020 for Tunisia (8.0 Mt in 2000) and Syria (3.9 Mt in 2006) represented less than 3% and 2% of global PR production (in 2020), respectively (data based on annual reports of Jasinski, (2002-2021).

Phosphate rock production in Morocco between 2000 and 2020 has steadily increased from 22 to 37 Mt year^{-1} (Jasinski, 2002-2021). However, the controversy surrounding PR mining in Western Sahara by Moroccan mining operations is widely documented (Chernoff and Orris, 2002; Leite et al., 2006; Arts and Leite, 2007; Mundy and Zunes, 2010; Boukhars and Roussellier, 2014; Camprubí, 2015; White, 2015; Smith, 2015; Hagen, 2015; Allan, 2016; Kingsbury, 2018; Omar, 2018). A summary of issues associated with the political context of Western Sahara is given in Focus Box 2.1.

Focus Box 2.1 - The conflict in Western Sahara

The ongoing conflict in Western Sahara is one of the more intractable legacies of European colonisation (Boukhars and Roussellier, 2014). Following the withdrawal of Spain in 1975, forces from Morocco and Mauritania moved in to occupy much of the territory. In 1975, the International Court of Justice issued a landmark ruling (The International Court of Justice, 2017)[i], that found no convincing historical evidence that Western Sahara belonged to anyone but the indigenous Sahrawi inhabitants.

The territory has a native population of less than half a million ethnic Sahrawis, 170,000 of whom have lived as refugees in Algeria since 1976 (UNHCR, 2018). In 1976, the Frente POLISARIO[ii] (also known as the Polisario Front) in the representation of Sahrawi, proclaimed the Sahrawi Arab Democratic Republic (SADR) as a sovereign State over the Territory of Western Sahara (Omar, 2018). Boukhars and Roussellier (2014) explain that this territorial dispute escalated into a war of independence between the POLISARIO, who were backed by Algeria, and the states of Mauritania (who withdrew in 1996), and Morocco who claim sovereignty.

It is widely accepted by legal scholars and under international law that Morocco has no legal title to Western Sahara (Chernoff and Orris, 2002; Leite et al., 2006; Arts

[i] The International Court of Justice stated "the Court's conclusion was that the materials and information presented to it did not establish any tie of territorial sovereignty between the territory of Western Sahara and the Kingdom of Morocco or the Mauritanian entity" https://www.icj-cij.org/en/case/61

[ii] Frente Popular para la Liberación de Saguía el Hamra y de Río de Oro (Frente POLISARIO).

and Leite, 2007; Mundy and Zunes, 2010; Boukhars and Roussellier, 2014; Camprubí, 2015; Smith, 2015; Kingsbury, 2018; Omar, 2018). Similar positions have been taken by the African Union (2017), the European Union (High Court of Justice (England & Wales), 2016) and the United Nations General Assembly (1980). For example, the European Court of Justice has ruled that treaties between the EU and Morocco do not cover Western Sahara, as this is not considered to be Moroccan 'territory' within the meaning of those treaties (The European Court of Justice, 2016).

It follows that Morocco remains an occupying power subject to international obligations concerning the exploitation of natural resources in that territory. These obligations prevent Morocco from exploiting these resources for its own benefit. By the same token, Morocco is constrained in granting exploitation licences to state-owned foreign companies. In a letter from the United Nations Under-Secretary-General for Legal Affairs and the Legal Counsel to the President of the Security Council in 2002 (in response to a request for legal advice), the UN Legal Counsel considered that it was not per se illegal for Morocco to conclude contracts with foreign companies to exploit mineral resources in Western Sahara, as has been done, but it would be illegal to do so if this was done 'in disregard of the needs and interests of the people of that Territory" (UN Legal Counsel, 2002).

A long-term solution to the issue does not appear to be in sight. In 1991, the United Nations Mission for the Referendum in Western Sahara (MINURSO) was established under the 'Security Council resolution 690 (1991) [Western Sahara]' settlement plan (United Nations Security Council, 1991). MINURSO was mandated to monitor a cease-fire and to organise an independence referendum. To date (2021) a referendum has not been held.

The Sahrawi Arab Democratic Republic (SADR), whilst not admitted into the United Nations, by 2021 has been recognised at some point in time by 84 of its member states and South Ossetia (45 of which have "suspended", "frozen" or "withdrawn" recognition)[i]. The African Union recognises both Morocco and Western Sahara as full member states. The United Nations observe Western Sahara as the last 'non-self-governing territory' in Africa (United Nations, 2020). Recent opinions of the European Court of Justice have reinforced that so long as Western Sahara is denied its right to self-determination, it is a non-self-governing territory (Court of Justice of the European Union, 2018). In 2020, in a change from 45 years of US policy, the Trump administration announced US recognition of Moroccan sovereignty over Western Sahara (Mundy, 2020) and this policy does not seem set for reversal under the Biden administration (Kasraoui, 2021).

[i] A full list of UN member states that have recognised the Sahrawi Arab Democratic Republic from 1975 to 2020, is provided on Wikipedia with relevant references included: https://en.wikipedia.org/wiki/International_recognition_of_the_Sahrawi_Arab_Democratic_Republic

Despite major interruptions of supply due to the Morocco-Polisario war (1976-1991), between 1976 and 2015, Moroccan investment in PR mining in the town of Bou Craa in Western Sahara has resulted in the export of over US$4 billion worth of PR (Smith, 2015). The Bou Craa reserves are mined by 'Phosphates de Boucraa S.A.' (Phosboucraa), a fully-owned subsidiary of the Moroccan state-owned phosphate mining company OCP. Phosboucraa also markets the PR from the Bou Craa mine and operates the loading dock and treatment plant located on the Atlantic coast at El Aaiún (or Laayoune). Whilst the Bou Craa mines represent 2% of Morocco's PR reserves (OCP, 2021), in 2016, 22% of Morocco's exported PR departed from El Aaiún (OCP SA, 2018). The average exports from Bou Craa over the last years have generated an annual income of around US$200 million (OCP, 2021). Phosboucraa is engaged in a development programme of approximately US$2.2 billion to move its operations up the value chain from raw materials to intermediate products and phosphate fertilisers by 2022 (OCP, 2015). OCP claim Phosboucraa is a major provider of economic viability and well-being of the region's inhabitants (OCP, 2015). Phosboucraa has around 2,195 employees, of which 75% are "locals" (without specifying whether they are Sahrawis or settlers). Zunes (2015) argues "the benefits of such 'development' have largely gone to Moroccan settlers and occupation authorities, not the indigenous population".

Disruption to supply from Bou Craa will not impact global P access significantly since around 1-2% of annual global PR production is in Western Sahara (based on PR production rate in 2019; see Jasinski, 2021). However, Camprubí (2015) argues that such quantities are significant enough to disrupt prices, and that countries currently and historically involved in exploiting the PR reserves in Western Sahara are not only interested in mining the resources, but also preventing others from doing so. Moreover, ethical questions for countries receiving PR supply from Bou Craa have been raised by several authors (including: Chernoff and Orris, 2002; Leite et al., 2006; Arts and Leite, 2007; Mundy and Zunes, 2010; Boukhars and Roussellier, 2014; Camprubí, 2015; Smith, 2015; Kingsbury, 2018; Omar, 2018). This controversy has been acknowledged by international courts (United Nations Security Council, 2002; Leite et al., 2006; White, 2015). For example, in December 2016, the Court of Justice of the European Union issued its decision in the appeal case of the POLISARIO against the European Council, concerning the EU's free trade arrangements with Morocco in Western Sahara (The European Court of Justice, 2016). The Court of Justice ruled that Morocco had no territorial right to make agreements covering Western Sahara with respect to free trade in this context.

Furthermore, ships carrying Western Saharan PR were detained in South Africa (OCP, 2018; WSRW, 2018) and Panama in 2017, with local courts asked to rule on the legality of their cargo. The High Court in South Africa ruled that the vessel NM Cherry Blossom carried PR that had been illegally exported from the territory, and was owned by SADR (High Court of South Africa, 2017). Conversely, the court in Panama ruled it did not have the right to hear the case, and the vessel was able to continue on its journey (Reuters, 2017a, b; Dudley, 2018).

Such concerns have not only been a matter of international dispute over resources but have also affected investment in PR production and import (Hagen, 2015; Allan, 2016; WSRW, 2020). The Western Sahara Resource Watch (WSRW)[i] report that the Danske Bank, the Norwegian Government Pension Fund, Luxembourg Pension Fund, KLP (the Norwegian insurance company), MP Pension (Denmark), the Norwegian Pension Fund, AP Fonden (the Swedish government pension fund), Nykredit Realkredit Group (Denmark) and PGB Pensioenfonds (the Netherlands), are among those to divest shares from OCP, and/or companies importing PR from Western Sahara (WSRW, 2018, 2020). They claim reasons for divestment[ii] are based on concerns of human rights breaches, international law violations and political controversy (WSRW, 2020). In 2018, the Canadian company Nutrien stopped importing PR from Western Sahara, stating this decision was based on a restructuring of the company (Nutrien, 2019). The WSRW report this caused a 50% drop in exports from the Bou Craa port of El Aaiún between 2018 and 2019 (WSRW, 2020). Notwithstanding this, in 2019, companies in New Zealand, India, Brazil and China imported PR from Western Sahara (WSRW, 2020).

For the present report, we do not focus on the legitimacy or otherwise of any territorial claims (including claims over natural resources) by Morocco or the SADR over land and mineral rights, nor on the determination of the European Court of Justice or other courts in regarding the merits or otherwise of any claim. Our role as sustainability researchers is rather to draw attention to these issues as part of providing a comprehensive picture of the challenges for sustainable P management.

Importantly, we note that this topic has become taboo for certain stakeholders. This section of the report has been subject to intense scrutiny as part of a peer review process linked to the UNEP-affiliated Global Partnership on Nutrient Management. The comparison of stakeholder engagement in sustainability discussions around P and nitrogen is particularly informative. Discussion about P is extremely sensitive. For example, a Policy Brief led by SCOPE identifying P as an emerging issue for food production (Syers et al., 2011) was reported by SCOPE staff as being exceptionally difficult to negotiate. Later, a policy brief for UNEP, prepared by some of the present authors, proved equally difficult, taking three years to finalise what was essentially a two-page brief (Brownlie et al., 2017). Experience in developing the International Nitrogen Management System by some of the authors (e.g. Sutton et al. 2021) has clearly shown that, if fast progress should be achieved with nitrogen, then P should be excluded from the stakeholder and intergovernmental discussion.

It is difficult to apportion the relative contribution of possible reasons for this extreme sensitivity about P, which we term here ‘phosphorus hypersensitivity’. For example, is P sensitive to producers and relevant industry associations because of the way that any public discussion might

i The Western Sahara Resource Watch (WSRW) is an independent international non-governmental organisation that works “in solidarity with the people of Western Sahara, researching and campaigning against Morocco’s resource exploitation of the territory”.
ii The WSRW provides quotes from divesting companies justifying their decisions to divest shares from OCP, and/or companies importing phosphates from Western Sahara on page 5 of WSRW, (2020) – www.wsrw.org/files/dated/2020-02-24/p_for_plunder_2020-web.pdf

affect resource value and market confidence of a commodity that has already been seen to be vulnerable to price instability? (See Figure 2.12 and Challenge 2.4 below). Or are the geopolitical interactions between companies and the relevant governments more important? Here it is possible to envisage that certain company interests would not wish to be in conflict with specific governments controlling access to PR reserves. In both cases, it can easily be seen how conflict surrounding PR in Western Sahara could spill over to make the whole topic of P extremely sensitive for certain stakeholders.

For this report, it is therefore not our purpose either to justify or to defend any party. Rather, we draw attention to the critical message from a global perspective: that P hypersensitivity associated with such geopolitical conflicts makes P difficult to discuss in the international arena. Consequently, it becomes even harder to make effective progress towards sustainable P management. Much more work is needed to better understand the complex geopolitical-business-sustainability dynamics at play. Future efforts should focus on finding governance models which, while considering the complexities involved, would enable relevant stakeholders to address the underlying issues and provide the confidence needed to accelerate the global conversation about phosphorus. Ultimately, this will be essential to mobilise the global adoption of sustainable P practices.

Challenge 2.4: Phosphate rock price spikes remain an ongoing risk

In 2008, phosphate rock prices spiked by 800%, causing a subsequent increase in fertiliser prices that affected the livelihood of many of the world's poorest farmers. This price spike occurred in response to a combination of factors, including instability in energy prices, changing dynamics of supply/demand for agricultural and phosphorus products, and the influence of geopolitics on exports. The stability of phosphate rock prices remains vulnerable to such drivers.

Two major PR price spikes have been observed in the last 30 years, in 1974 and 2008 (Rosemarin and Ekane, 2016) (Figure 2.12). Mew (2016) predicted that PR price spikes should be expected in the future. After the 1974 price spike, the price of PR remained reasonably stable at around US\$50 t^{-1}. In 2008, PR prices increased by 800%, impacting P fertiliser prices accordingly (de Ridder et al., 2012), with a further modest spike in 2012. Spiralling fertiliser prices eventually led to a crash in demand for P, and the PR price dropped. However, as observed by Mew (2016), although the PR price dropped after each price spike, it has dropped to double the baseline preceding the increase (Figure 2.12).

The cause of the 2008 PR price spike is complex and likely driven by a combination of factors including

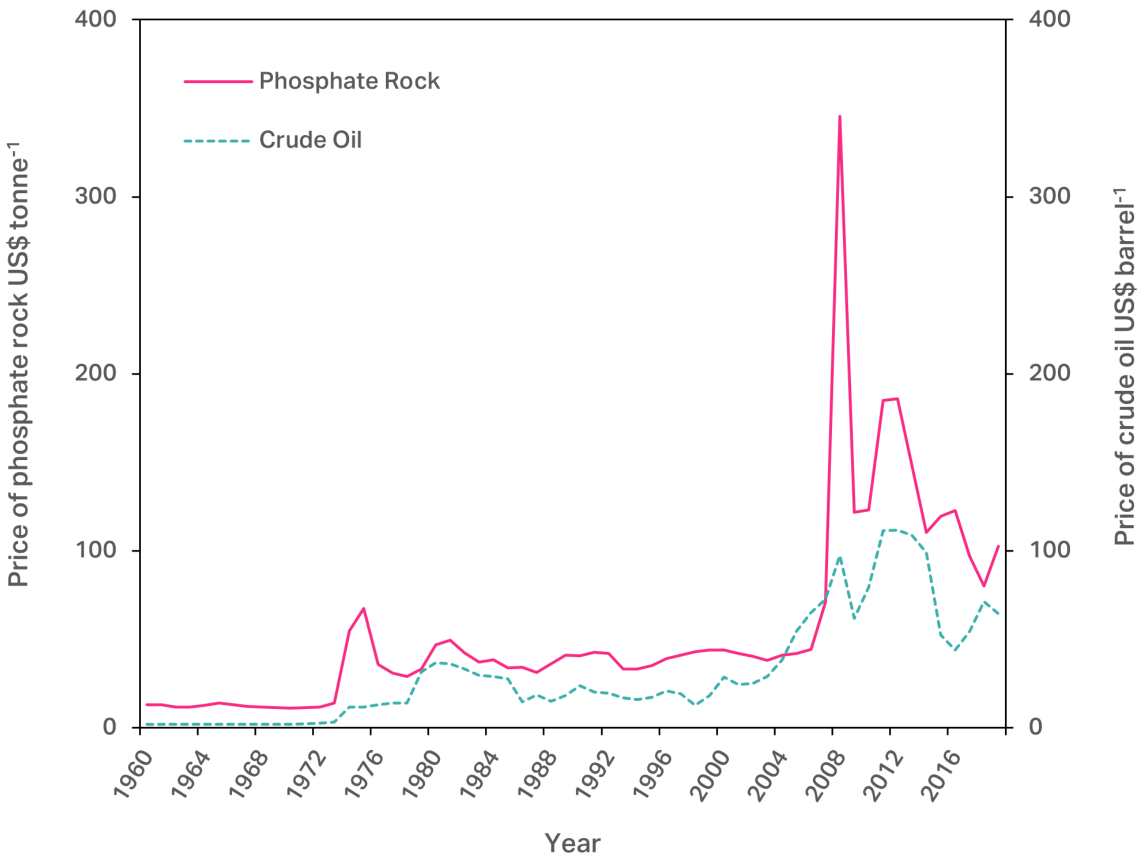

Figure 2.12 Phosphate rock (PR) and crude oil prices 1960–2019, indicating a PR price spike in 1974 and an 800% price spike in 2008, with another spike following in 2012. After each price spike, the cost drops and stabilises at a higher cost. Other commodities experienced price spikes in 2008, including crude oil. Data source: World Bank Commodity Price data.

changing the market supply/demand dynamics for agricultural and P products, instability in energy prices and geopolitical control on exports. Indeed, the price spike in 2008 was not unique to PR and affected almost all commodities, including oil, minerals, and grains (Martin and Anderson, 2012). An International Fertiliser Association (IFA) bulletin suggested the 2008 volatility in agricultural and fertiliser prices reflected a combination of long- and short-term factors (IFA, 2011). Long-term factors included:

- income growth and dietary changes in emerging economies;
- competing objectives for agriculture, in addition to food and feed production (e.g. production of fibre, biofuel feedstock and bio-chemicals);
- global grain market conditions (i.e. low cereal stocks); and
- regular increases in energy prices.

Contributing short-term factors included economic weakness in many countries, export restriction measures and extreme weather conditions and natural disasters (IFA, 2011). Some reports suggested the introduction of a US ethanol policy and increases in food prices were the major contributors to the increases in P demand (Cordell et al., 2009; Childers et al., 2011). However, Khabarov and

Obersteiner (2017) argue that fertiliser market policies in India, which led to India doubling its import of P-fertiliser in 2008 at a time when prices doubled, were the major contributor to the global PR price spike.

The 2008 price spike affected the livelihood of many of the world's poorest farmers, resulting in farmer debt and reports of farmer riots, for example, in Haiti and India (Cordell and White, 2014). Mew (2016) states "these price spike events can be seen to be related to the escalating investment cost required by new mine capacity, and as such can be expected to be repeated in future…[and] …phosphate rock price volatility is likely to have more impact on food prices than rising phosphate rock production costs." In 2021, the PR price began to rise steeply again.

Challenge 2.5: There is a lack of transparent, complete and comparable phosphate rock data

Significant discrepancies in phosphate rock data are reported, making it difficult to assess accurately the risk of geographic depletion of reserves. Differing definitions for phosphate rock 'reserve' and 'resource' are a cause of discrepancies. Datasets on phosphate rock reserves and resources are commercially sensitive and are often not publicly available. Reserve estimates are dynamic and require regular updating, while conformity in data and reporting is needed. The United States Geological Survey estimates global phosphate rock reserves in 2020 at 70,000 Mt, indicating a current lifetime of >300 years, although a longer lifetime may be expected in practice due to innovation and price elasticity.

The current rate of PR mining far outweighs the geological replacement of reserves and resources. Geological deposits of PR take millions of years to form, such as via the decomposition of marine organisms. In 2020, 223 Mt of PR (containing 67 Mt P_2O_5) were mined from global PR reserves estimated at 71,000 Mt of PR (containing 21,300 Mt of P_2O_5) (Jasinski, 2021). In the same year, 47 Mt of P_2O_5 was consumed in the anthropogenic global P cycle, in phosphoric acids, fertilisers and other products (Jasinski, 2021). Even if only currently-identified reserves are considered (excluding P resources which are a factor of 4 larger), this indicates that there is little risk of scarcity of PR in the coming decades (Ulrich and Frossard, 2014). However, access to P for many smallholder farmers, especially in parts of Africa, remains a significant issue, impacted by factors other than geological scarcity associated with total P reserves and resources (Cordell and White, 2014) (see Chapter 3).

Concerns over the depletion of PR reserves are not only a contemporary issue (Ulrich and Frossard, 2014), as illustrated on the front cover of 'The Fertiliser Review' from 1938, which depicts concern in North America over depletion of reserves (i.e. Eastern Field) against a backdrop of considerable 'resources' (i.e. Western Field) (Figure 2.13).

In 2009, there were renewed concerns about the geological depletion of P reserves under the term 'Peak Phosphorus' (Déry and Anderson, 2007) resulting in contention in accounting reserves (Déry and Anderson, 2007; Cordell et al., 2009) and debates on how long PR reserves may last (e.g. Cordell and White, 2011; Scholz et al., 2013; Ulrich and Frossard, 2014; Geissler et al., 2018). As Ulrich and Frossard (2014) explain, however, concerns about P scarcity, when framed as geologic depletion alone, have been repeatedly disproven by resource appraisals from industry or government experts.

Authors who have conducted estimates in the past share frustration that PR reserve data are unsatisfactory and call for conformity in reporting (Brobst and Pratt, 1973; McKelvey, 1974; Bender, 1986). Currently, estimates are constrained by a lack of available information and

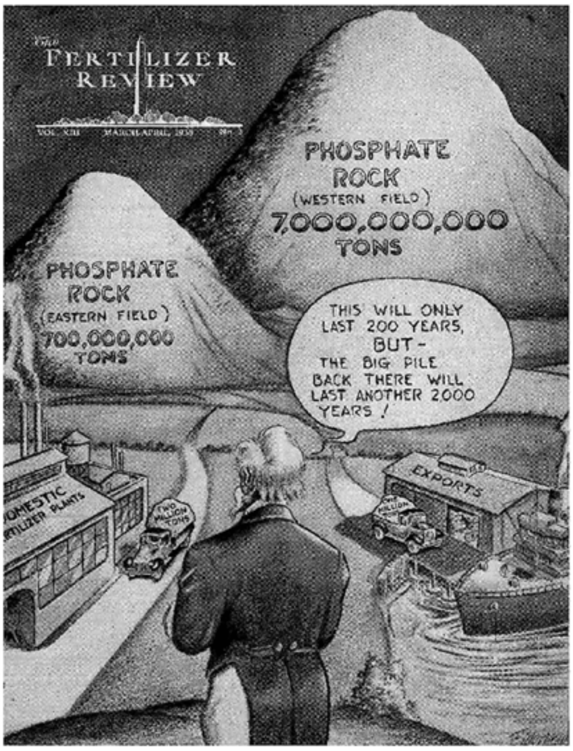

Figure 2.13 Cover of The Fertiliser Review Vol. XIII, No. 2, March–April 1938. The small pile of phosphate rock (PR) represents 200 years left of supplies in Florida, with the larger pile representing 2000 years left of other US PR resources, illustrating the role of the undeveloped western PR deposits in USA phosphorus supply considerations (image featured in Ulrich, 2016).

often based on data collected using different methodologies (Ulrich and Frossard, 2014). Notable publicly available datasets include:

- USGS PR reports (the major data source for global scientific publications on mineral resources) (https://www.usgs.gov/centers/nmic/phosphate-rock-statistics-and-information).
- FAO fertiliser datasets, available at its FAOSTAT data portal (http://www.fao.org/faostat/en/).
- IFA data portal (IFASTAT) (http://ifastat.org) which provides limited publicly available data, with extensive data for members.

- World Bank 'pink sheets' (https://www.worldbank.org/en/research/commodity-markets) providing long-term price series for PR and various mineral fertiliser types.

However, since estimates are not reported in the same units (i.e. P_2O_5 or P content, or amount of PR) comparable analyses have to rely on assumptions that may not be accurate (e.g. concentration of P in PR, which differs between rock type). There is no independent source of data or governance at the international level to set standards and norms for reporting on PR (Rosemarin and Ekane, 2016). A review of the commonly used data sources on PR by Geissler et al. (2018a) identified current and significant discrepancies in global PR production data.

A major source of discrepancies has been variation in the definitions of PR 'reserve' and 'resource' (Nedelciu et al., 2020). Phosphate rock 'reserves' can be considered PR resources that can be economically produced. However, 'economic' is contextual, dependent on changing socio-economic conditions and technological capacity. Currently, a classification method for monitoring reserves is not universally agreed upon, or required by law (Edixhoven et al., 2014), which can lead to uncertainty in reserve and resource speculations.

For example, in 2010, reserves and resources as reported by the USGS were reassessed by the International Fertiliser Development Centre (IFDC). The USGS reported the global PR reserves in 2010 of 16,000 Mt, 36% of which were in Morocco and Western Sahara. In the same year, the IFDC issued a report (van Kauwenbergh, 2010) in which global PR reserves were estimated at 60,000 Mt, with 85% in Morocco and Western Sahara. In the USGS report, reserves were defined as an upgraded concentrate (when the information is available), a term used by the mining industry for ore processed to remove tailings to give a concentrate with phosphate content of at least 30%, ready for sale on the market. In the IFDC report (van Kauwenbergh, 2010), the term 'PR deposit' is used for both unprocessed rock and beneficiated concentrates, although it is not detailed how the weight of beneficiated ore is converted to the weight of PR. The USGS subsequently adopted the reserve estimation of the IFDC (de Boer et al., 2019), resulting in significant increases in the estimated PR reserves from 2009 to 2010 in Morocco and Western Sahara and in the Middle East (Figure 2.14).

Reserves estimates are dynamic and require regular updating. Currently, there is a high degree of uncertainty on the extent of PR reserves (Edixhoven et al., 2014). As with oil reserves, the effort to discover new reserves is not static and will increase with impending shortages, and technological development. Furthermore, some PR resources may become economically profitable to exploit as prices of PR rise or through innovation to increase P yield or reduce processing costs.

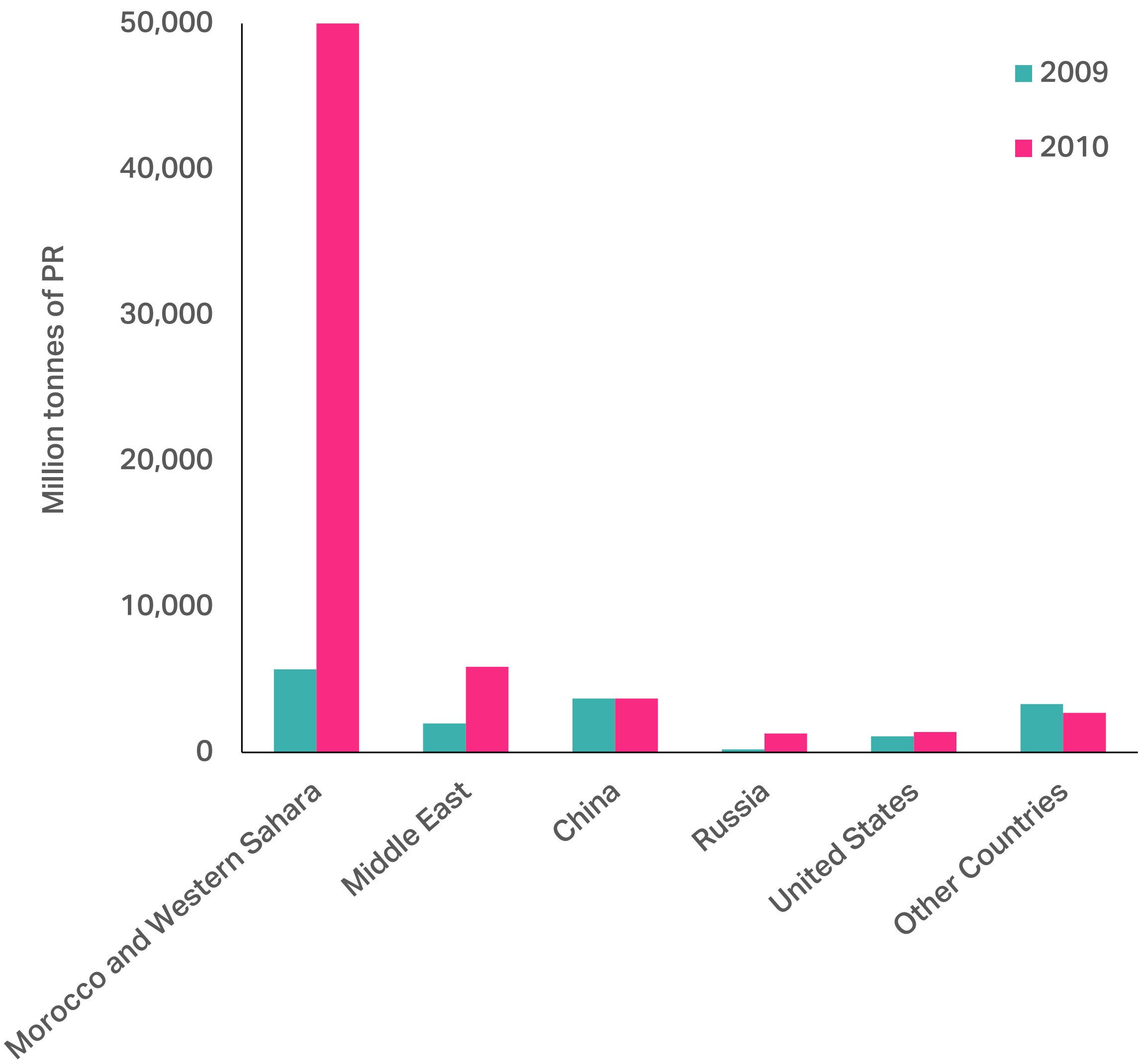

Figure 2.14 Estimated world reserves of Phosphate Rock (PR) in Mt in 2009 and 2010, according to USGS (Jasinski, 2009). In 2011, following a reassessment of PR reserves by the International Fertiliser Development Centre (van Kauwenbergh, 2010), the USGS adopted a different definition for 'PR reserve' (as used by the IFDC). This resulted in the USGS reporting a significant increase in the estimated size of PR reserves in Morocco and Western Sahara, from 5700 Mt in 2009 to 50,000 Mt in 2010, despite no new PR reserves being identified (compare with Figure 2.8). The definition of 'PR reserve' has a significant impact on PR reserve reports and our understanding of P scarcity.

2.6. Solutions

Solution 2.1: Reduce reliance on mineral phosphorus fertiliser

Replacing mineral phosphorus fertiliser with recycled phosphorus fertiliser would help to shift reliance away from mined phosphorus sources. Optimising capacity to recycle phosphorus throughout the food value chain in combination with societal change (e.g. diet change) would help to reduce phosphorus demand and losses. Enabling mainstream production of sustainable recycled phosphorus fertilisers containing low concentrations of contaminants is an essential prerequisite to upscaling operational recycling.

National and international resilience to mineral P fertiliser supply risk can be increased by implementing a more circular approach to P use (Withers et al., 2015a). A framework to address inefficient P use throughout the food value chain is outlined in the 5R stewardship framework (Withers et al., 2015a). The 5R framework promotes Re-aligning P inputs, Reducing P losses, Recycling P in bioresources, Recovering P in wastes, and Redefining P in food systems. Through consideration of these elements, it can help identify and deliver a range of integrated, cost-effective and feasible technological innovations to improve Phosphorus Use Efficiency (PUE, defined and discussed in Chapter 4) in society and reduce dependence on mined P imports (Withers et al., 2015a). For most regions, this will require a combination of actions highlighted in later chapters in this report, including significantly improving PUE in agriculture (Chapter 4), reducing societal P demand (Chapter 8), and shifting reliance from PR to recycled P from both domestic and international sources (Chapters 6 and 7).

National policies that optimise P recycling, and hence reduce reliance on mineral P fertilisers, are acknowledged as pivotal to a transition to a more sustainable P future (Koppelaar and Weikard, 2013; Reijnders, 2014; Cordell and White, 2014; Withers et al., 2015b; van Dijk et al., 2016; Chowdhury et al., 2017; Withers et al., 2018a; van Kernebeek et al., 2018). Switzerland, Sweden, Austria and Germany are pioneering policies to recover P from waste streams (Günther et al., 2018). Furthermore, the revision of the Fertilisers Regulation (EC) No 2003/2003, aimed to increase market opportunities for recycled P fertilisers to give farmers and consumers a wider choice of more sustainable products, promote green innovation and help to develop the circular economy while reducing dependence on imported nutrients (European Parliament, 2018).

Currently, the use of recovered P to produce fertiliser cannot compete financially against phosphate rock. To make a significant increase in the use of the recovered P, the mineral fertiliser industry must make a substantial shift to increase its use of recovered P as a raw material (see Chapter 7). Increasing national capacity and further research and development to support mainstreaming of fertiliser production using recovered P materials is essential as a foundation to reduce costs, although issues of contaminants should also be addressed (Kratz et al., 2016; Kumpiene et al., 2016; Bigalke et al., 2017) (see Chapters 6 and 7). Marketing opportunities for using recycled P sources may also merit consideration by the fertiliser industry. For example, a goal for fertiliser products to contain a minimum of 20% recycled P by 2030 could set a benchmark that demonstrates green commitment across the industry.

Solution 2.2: Establish safety levels for contaminants in fertilisers and agricultural products

Internationally agreed limits should be set for cadmium and harmful contaminants in mineral and recycled phosphorus fertilisers and food. Existing national cadmium limits require better enforcement. Optimising fertiliser use to match plant needs and practices to reduce phosphorus losses can also decrease inputs, thereby further lowering the application of fertiliser contaminants to soils, complementing the use of clean mineral and recycled phosphorus fertilisers.

Implementing safe limits for cadmium and other potentially harmful contaminants in all P fertilisers and feed supplements should be applied globally, especially when considering the global trade in agricultural produce (Bigalke et al., 2017; Hooser, 2018). National limits on cadmium in mineral P fertilisers exist in 21 EU countries and to a lesser extent around the world (Ulrich, 2019), though Bigalke et al. (2017a) argue that existing cadmium limits need to be better enforced. In addition to reducing contaminant inputs, a reduction in the recycling of contaminants in manure and other organic resources may be necessary. Optimising fertiliser practices to match plant needs, reducing fertiliser losses and increasing the use of clean and high-quality mineral and recycled P fertilisers are all important in reducing the load of fertiliser-borne contaminants to the environment (Bigalke et al., 2017; Hermann et al., 2018). Furthermore, contaminants can be removed during PR processing and fertiliser production through existing measures such as blending (mixing of PRs with varying levels of cadmium) and decadmiation (i.e. processes to remove cadmium) (Syers, 2001; Benredjem and Delimi, 2009). However, such processes can be expensive and, given limited regulatory requirements in many territories, they are not currently used at industry scales.

Whilst there has long been a lack of global policy or regional legislation on cadmium levels in fertiliser products, this is starting to change. In 2019, the European Parliament and Council adopted Regulation 2019/1009 which set harmonised cadmium limits for CE marked phosphate fertilisers (European Parliament & the Council of the European Union, 2019). Whilst cadmium limits are currently unrestricted, by 2022 P fertilisers must contain less than 60 mg cadmium kg^{-1} P_2O_5 (European Parliament & the Council of the European Union, 2019). Further tightening of this limit to 40 mg kg^{-1} (by 2025) and 20 mg kg^{-1} over the next 12-16 years has been proposed (Ulrich, 2019), but was not accepted by the European Commission.

Solution 2.3: Promote models of governance aimed at ensuring phosphorus security

Ensuring phosphate security which supports all farmers to access sufficient phosphorus to grow crops, is a global responsibility and requires international cooperation. Balanced stakeholder participation in negotiations is necessary to ensure phosphate security and avoid domination of regulatory agencies by industries or private interests. An internationally agreed framework promoting sustainable phosphate rock mining and trading is currently missing and urgently needed.

Ensuring all farmers (in particular, on small-scale farms in less economically developed countries) have access to sufficient P to grow crops and are buffered from fertiliser price fluctuations is a global responsibility and requires international cooperation (Teah and Onuki, 2017). There are several dimensions to this from a legal perspective. Under international law, states are entitled to control the natural resources located in their territory and maritime areas. This is an attribute of their sovereignty and stems from the well-recognised principle of permanent sovereignty over natural resources. As such, states are entitled to freely decide whether to exploit their resources (even, if they wish, to the point of depletion) or instead to leave them in situ. In light of the unequal distribution of PR reserves and the growing need for all states to have access to P, it would be sensible for states to explore models of governance driven by a benefit-sharing approach, following examples adopted for other natural resources (De Jonge, 2011; Morgera, 2016).

Two other areas of international law are also relevant. First, international trade obligations prohibit export bans (but not production bans) on products, unless this is justified on good grounds, e.g. environmental considerations. Second, human rights law requires states, in certain circumstances, not to harm persons in other states. In this respect, it is relevant that access to P is an essential step towards reaching the objectives of SDG 2 - Zero Hunger (Gil et al., 2019; El Wali et al., 2021). The right to food is recognised in the 1948 Universal Declaration of Human Rights as part of the right to an adequate standard of living and is enshrined in the 1966 International Covenant on Economic, Social and Cultural Rights (United Nations, 2010). This does not mean that states with P are necessarily under an obligation to allow exports of P to states in need of fertiliser, but it is a relevant consideration in any negotiated solution, and also plays a role in the interpretation of other relevant international rules.

Balanced stakeholder participation in negotiations is necessary to ensure P security and avoid 'regulatory capture' by industry. The concept of 'regulatory capture' has been introduced to reflect situations where the decisions of regulatory agencies are dominated by the industries or interests they are charged with regulating (De Jonge, 2011). There is also a need to recognise socio-political power differences between different stakeholders, both nationally and internationally. This may require engagement to go beyond governments and international organisations to include indigenous communities and their respective rights over specific resources, including healthy ecosystems, access to agricultural lands and access to clean water (De Jonge, 2011; de Ridder et al., 2012; Smith, 2015). Indeed, PR mining can require large amounts of water, which may challenge the needs of local communities. Furthermore, potential adverse environmental effects and human health risks associated with the discharge of phosphate mining waters have been reported (Chraiti et al., 2016; Reta et al., 2019). It is appropriate that assessment of social impacts be informed through consultation with all stakeholders, and careful analysis of the complex relationships between all stakeholders involved in the mining, trade and use of P from PR reserves (Nedelciu et al., 2020). Two initiatives in the mining sector have relevance here: the Extractive Industries Transparency Initiative (EITI) and the Due Diligence Guidance for Responsible Mineral Supply Chains from Conflict-Affected and High-Risk Areas (DDG) of the Organisation for Economic Co-operation and Development (OECD). The EITI Standard (EITI, 2019) requires the disclosure of information along the extractive industry value chain, such as how extraction rights are awarded, how revenues make their way through the government and how these revenues benefit the public. Whilst currently implemented in 55 countries, it would be appropriate for the EITI standard to be extended to all nations that mine PR reserves. The OECD DDG provides detailed recommendations to help companies respect human rights. The OECD DDG cultivates transparent mineral supply chains and sustainable corporate engagement in the mineral sector. It aims at enabling countries to benefit from their mineral resources, whilst preventing the extraction and trade of minerals from becoming a source of conflict, human rights abuses, and insecurity. The OECD DDG is available for use by any company potentially sourcing minerals or metals from conflict-affected and high-risk areas (OECD, 2016).

Ultimately, it is up to the international community to agree on the exact terms and provisions of a fair and equitable PR sharing mechanism. The World Trade Organization and the United Nations are perhaps best positioned to take a leading role in facilitating these processes. It is recommended that states should seek to adopt a multilateral legal framework promoting 'sustainable PR mining and trading'. Although PR mining cannot be fully sustainable, as it depletes a natural resource, it can be optimised and made more efficient (Nedelciu et al., 2020). Such a framework would need to consider environmental impacts, socioeconomics, fairness for local communities and intergenerational equity. Its activities would need to consider reasonable export/import taxation of PR and trade agreements that are not exploitive (e.g. addressing potential concerns about favouring rich countries and affecting access of poorer nations to P reserves).

It is recommended that progress in P sustainability would be advanced by the establishment of an appropriate United Nations body entrusted with making a global assessment of all PR reserves (Rosemarin and Ekane, 2016; Nedelciu et al., 2020) and developing harmonised criteria for 'sustainable PR mining and trading'. A potential framework to address these needs is suggested in Chapter 9, and in Brownlie et al. (2021). Significant investments and compromises from all governments and other stakeholders are needed to move forward successfully. The approach of addressing common but differentiated responsibilities as used in climate change negotiations could provide a suitable model. In this way, the parties with more means (e.g. high-income countries) might take the lead in promoting PR security and equitable access to phosphorus.

Solution 2.4: Improve stakeholder capacity to deal with phosphate rock price volatility

Stakeholders need to plan for uncertainty by increasing adaptive capacity. Building national capacity to close the phosphorus loop in food production systems and shifting reliance from mined phosphorus to recycled phosphorus will help protect against phosphorus supply risk. Governments need to recognise phosphorus supply risks through appropriate policy and regulation.

Understanding the drivers of PR price spikes is important as a foundation to anticipate and mitigate the impacts of future volatility. Given PR price spikes are expected in the future (Mew, 2016), P stakeholders need to plan for uncertainty by increasing adaptive capacity. Ultimately, improving nutrient management to ensure the most efficient use of P in agriculture (as discussed in Chapter 5), and improving circularity in national P cycles will provide the greatest protection. Progress towards this goal requires a shift in reliance away from mined P sources to recycled P sources and the reuse of organic sources of P on farmland (see Chapters 6 and 7). Implementing strategies to optimise P demand and investing in innovations to access legacy P from soil stores and reduce P losses to the environment are also required. However, for most nations 'phosphate independence' is unrealistic, even impossible; governments, therefore, need to recognise P supply risks within national policy. To assess supply risk, governments and stakeholders require accurate and comparable data on reserves, resources, supply and demand at the national scale. Approaches have been developed to produce these data to inform analyses of P supply resilience (Rosemarin and Ekane, 2016; Geissler et al., 2018).

Governments need to recognise P supply risks through appropriate policy and regulation. The 2008 fertiliser price spike was certainly a trigger for the European Union to place PR on the EU list of critical raw materials (European Commission, 2014), which has in part led to innovative legislation in Germany and Switzerland to increase P recycling from wastes. Whilst current processes to recover P and manufacture recovered P fertilisers can be expensive (see Chapter 7), legislation to support innovation and growth in P recycling industries can help to bring down prices of recycled P fertilisers.

Finally, the international financial sector can play a crucial role in shaping the behaviours of the industry with respect to reducing the financial impact of P losses through the food system and in realising the financial benefits of new investment in companies developing sustainable P approaches. There is an opportunity to dovetail and build upon the progress currently being made in climate change responses across the international financial sector, recognising that unsustainable nutrient use is a key driver of climate change and environmental degradation. These impacts carry large financial costs, for example, to the drinking water sector, whereas the emergence of more sustainable nutrient use technologies and approaches (e.g. the recycling industry) represents an opportunity for future investment. An example of the type of approach that could be utilised is the Task Force on Climate-related Financial Disclosure (TCFD, 2017), which was established by the Financial Stability Board of the G20 group of countries. Nutrient-related financial information could be developed to help support investors and companies make evidence-based decisions on their response to environmental change, new regulations, and customer behaviour associated with a transition towards a more sustainable P future.

Solution 2.5: Improve transparency and the independent assessment of phosphate rock data

There is a need for transparency and free access to accurate, current data on global reserves and resources of phosphate rock. An independent, international body is needed to assess data regularly and to disseminate findings through appropriate mechanisms, institutions, and outreach programmes.

The need for transparency on global reserves and resources of PR and estimated quality of supply is recognised widely (e.g. Ulrich and Frossard, 2014; Rosemarin and Ekane, 2016; Wellmer and Scholz, 2017; Geissler et al., 2018; Nedelciu et al., 2020). Increasing transparency will require the collaborative efforts of governments, industry stakeholders, geological surveys, academia and inter/-national organisations (van Kauwenbergh, 2010; Ulrich and Frossard, 2014).

Strategies for P management often rely on quantitative models to underpin decision-making and policy development. Independent of the modelling approach, robust data is essential. With the lack of available data, decisions taken by governments may not be correct, and predictions by scientists uncertain (Geissler et al., 2018). Currently, private consulting firms collect data, which are then available to paying members. For example, CRU is a business intelligence company that collects data (e.g. mineral P fertiliser production costs, prices, supply and demand) and uses IFA data to cross-check its numbers, whilst scientists usually depend on access to publicly available data from national geological surveys. Geissler et al. (2018) suggest IFA data on global PR production is likely to be the most accurate, and covers 98% of global production, but points out that only limited reports and data are available to non-members.

The establishment of an independent and international body to regularly assess data and to promote and disseminate findings/results through appropriate mechanisms, institutions and outreach programmes is urgently needed (Rosemarin and Ekane, 2016; Geissler et al., 2018; Nedelciu et al., 2020; Brownlie et al., 2021) (see Chapter 9). This could serve a similar function for P supply and demand, as the UN World Water Quality Alliance does for water quality, by providing governments and other stakeholders with relevant evidence-based assessment, scenarios, solutions, and services. It is recommended that such activities be accompanied by efforts to agree on the definitions of reserves and deposits, and conversion of all phosphate data to a common base, such as 100% P_2O_5 ("total tonnes of P_2O_5 nutrients") (Geissler et al., 2018). In the meantime, policymakers should be made aware of the discrepancies and uncertainties in PR data when making policy decisions.

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03

Transforming food systems: implications for phosphorus

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Left: A woman sells fruits and vegetables at Siti Khatijah Market, Kelantan, Malaysia. 'Phosphorus security' envisages a world where all farmers have access to sufficient phosphorus to grow enough food to feed a growing population a healthy diet while ensuring farmer livelihoods and minimising detrimental environmental and social impacts. Photograph taken by Alex Hudson on www.unsplash.com - www.unsplash.com/@aliffhassan91

Managing phosphorus underpins the sustainability of the food system and is vital in achieving future food security. Strategies to deliver phosphorus sustainability include a transition to circular phosphorus value chains, land-use planning approaches that support greater phosphorus use efficiency and a reduction in consumption of animal products. Affordable access to sustainable phosphorus sources is imperative to ensure food provision for all and to protect the livelihoods of smallholder and marginal farmers.

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Challenge 3.1: Business as usual is unsustainable: we must produce healthier foods, using appropriate phosphorus inputs

Our food system is a significant cause of nutrient pollution in terrestrial, freshwater and marine ecosystems, and of global climate change, while more than half the global population are acutely hungry, malnourished, overweight, or obese. The public health and ecological costs of the current food system exceeds the economic value of agriculture. Systemic transformation is required for food systems to become environmentally sustainable and provide nutritional security for all. Sustainable phosphorus strategies must directly support, not hinder, this transformation. On the current path, the global food system will increase the mining of finite phosphate rock to produce fertiliser, feed additives and food supplements, and is not tracking towards a circular phosphorus system (driven on recycled phosphorus inputs).

Challenge 3.2: Increasing global consumption of animal products is increasing phosphorus demand

The amount of phosphorus required to produce the average per capita global diet has increased by 38% in the last 50 years, due to the rise in consumption of animal products, increase in average per capita consumption and increased food waste. Excluding phosphorus-efficient grass-based systems, a large proportion of cropland is needed to support intensive meat and dairy production through concentrated animal feeding operations. This trend is driving increased mining of phosphate rock for fertilisers, animal feed and supplements. Unhealthy diets, including overconsumption of animal products, are also a significant contributor to non-communicable diseases.

Challenge 3.3: Balancing intensive agriculture with low input farming

Agricultural intensification increases productivity yet increasing phosphorus inputs to crops can also over-enrich adjacent land and waterbodies with nutrients. Lowering phosphorus inputs reduces environmental risk and promotes biodiversity but may restrict yield in the long-term. Strategies need to provide the right balance of intensification to avoid the need to convert more land to agriculture. Optimising the multitude of costs and benefits and taking account of direct and indirect impacts can be challenging and context specific. The challenge we face is in developing low phosphorus input farming systems which can sustain food production.

Challenge 3.4: Many farmers lack access to phosphorus, threatening their livelihoods

Currently, 1 in 7 farmers cannot access or afford phosphorus fertilisers to increase productivity, reducing their ability to maintain food security and livelihoods. Those farmers most affected are rural smallholder farming families, particularly in less economically developed countries, but also in some more economically developed countries. There are marked global inequalities in access to phosphorus as a resource, leading to substantial inequalities in the distribution of risks to food security.

Solution 3.1: Managing phosphorus sustainably can support a shift to healthier diets

Global food systems must produce, actively support, and provide access to nutritious food and diets for all. This shift, from 'market-led' to 'sustainable' food security, can reduce phosphorus demand and adverse impacts on ecosystems and society. Concurrently, strategies to deliver better phosphorus sustainability, including circular phosphorus value chains, can benefit agricultural economies, whilst effective monitoring systems, data sharing, and knowledge exchange can ensure strategies adapt to a transforming food system.

Solution 3.2: Shift global consumption of animal products towards plant-based diets

Reduced consumption of animal products especially from intensive production systems in some regions may reduce global agricultural phosphorus demand and contribute to healthier environments. Increased awareness amongst policymakers and the public of the environmental impacts of phosphorus use in food production, and the human health risks of excessive consumption of animal products, will be an essential driver of change. Knowledge exchange between academics, stakeholders and the public can help identify solutions to support a transition to more phosphorus sustainable consumer behaviour, as could policy and regulatory changes (including internalising the environmental costs into food pricing).

Solution 3.3: Integrated landscape strategies to improve phosphorus use efficiency and reduce losses

There is an opportunity to develop novel land-use planning approaches to support more sustainable phosphorus use across multiple and interacting contexts. These include agricultural production, ecosystem and human health, local economies and regional capacity for institutional planning and coordination. Sustainable farming systems in which animal and crop production are more integrated and animal residues and manures are treated as valuable phosphorus resources, will support efforts to increase phosphorus use efficiency within landscapes while reducing negative impacts on aquatic and terrestrial ecosystems.

Solution 3.4: Better support for smallholder farmers

Affordable access to sustainable phosphorus sources is imperative to ensure food provision for all and to protect the livelihoods of smallholder and marginal farmers. Multiple options exist to help improve phosphorus access in these communities. These include access to credit, extension services, investment in sustainable infrastructure (such as local phosphorus recycling systems from food waste and sanitation where available), and knowledge exchange to support better phosphorus use efficiency and recycling. Developing the capacity to recycle phosphorus from local and regional food systems where available can help to shift reliance away from mineral phosphorus fertilisers.

3.1 Introduction

All farmers need access to phosphorus (P) to grow crops, regardless of what they grow or where they farm. Yet, access to affordable and sustainable sources of P is, currently, not guaranteed. At the same time, excess P use can harm aquatic ecosystems and in turn the food, ecosystem and agricultural services they support (see Chapter 5). Managing P sustainably, therefore, underpins the sustainability of the food system and is vital in achieving future food security. To make progress towards a sustainable global food system we must take a multiple-stressor mitigation approach. This includes better managing P use in addition to other essential macro- and micro-nutrient sources essential to food production to reduce the impact of food production on climate change, human and ecosystem health, and to addresses inequalities in access to nutritious food from local to global scales. Efforts focused on single-stressor action are not sustainable and will be unlikely to tackle the scale of the challenge. Multiple United Nations Sustainable Development Goals (SDGs) demand a transformation to more sustainable food systems, however, the role of P management in food systems is not yet sufficiently addressed. 'Phosphorus security' envisages a world where all farmers have access to sufficient P to grow enough food to feed a growing population a healthy diet while ensuring farmer livelihoods and minimising detrimental environmental and social impacts (Cordell, 2010).

Actions to improve global P security should be underpinned by a comprehensive understanding of food systems and the flows of P within them. Such actions should be co-developed with relevant stakeholders to achieve food security whilst delivering multiple benefits to society, for example as defined by the SDGs. This will decrease the likelihood of 'lock-ins', where actors are unwilling or unable to exit a position because of sunk infrastructure costs, regulations, or penalties. For example, while phosphorus recovery will be essential, investing in expensive phosphorus recovery technologies that do not produce phosphorus in biochemical forms that can be easily used to produce fertilisers, and are more energy-intensive to produce than mineral fertilisers, may not be the best use of financial resources.

One significant challenge to overcome with respect to the global food system is the equitable supply of, and access to, phosphorus. Only a few countries control the bulk of non-renewable phosphate reserves and production, due to natural geological phosphate formations. Most countries are import-dependent and hence vulnerable to price shocks, supply disruptions, and import barriers (see Chapter 2), which can disrupt food security and farmer livelihoods if not sufficiently managed.

In this chapter we provide an overview of the importance of sustainable P management in global food security, outlining the need to transform our food system to enhance public health and farmer livelihoods whilst reducing adverse impacts on the environment. We then propose possible future scenarios for food systems and highlight that whilst the future trajectory of food system transformations remains uncertain, it should embrace both trade-offs and synergies for P security. We then summarise some of the key challenges and solutions to achieving P security within a food system under transformation.

3.1.1 Food and fertiliser price spikes

Food and fertiliser prices have a significant impact on food systems, especially in developing economies. In 2007–2008, the nominal prices of almost all food commodities increased by more than 50%, then dropped soon after, and surged again in 2010–2012 (FAO, 2017). This was accompanied by an 800% increase in the price of phosphate rock in 2008, with a further peak in 2011–2012, impacting P fertiliser prices and food security (de Ridder et al., 2012) (Figure 3.1).

Against the background of declining food prices since the 1950s (Jacks, 2018), the 2007–2008 increase in the FAO Food Price Index came as a global surprise. It was likely driven by a combination of increasing biofuel demand, speculation in commodity futures markets, countries' aggressive food stockpiling policies, trade restrictions, macroeconomic shocks to the money supply, exchange rates, and economic growth (Tadasse et al., 2014). Whilst the 2008 phosphate rock (PR) price spike is connected, the drivers are again complex, contentiously debated, and likely include a combination of market supply and demand dynamics for agricultural products, instability in energy prices and geopolitical control on exports (Childers et al., 2011; Cordell et al., 2015; IFA, 2011; Khabarov and Obersteiner, 2017) (see Chapter 2).

Spikes in food and fertiliser prices directly affected access to food for the global poor and vulnerable. The risk of food production being unable to meet the rapidly growing global population demand for food (and the water and energy needed to supply it), as well as the impacts of climate change, was cause for global concern, placing P sustainability on the global food security agenda.

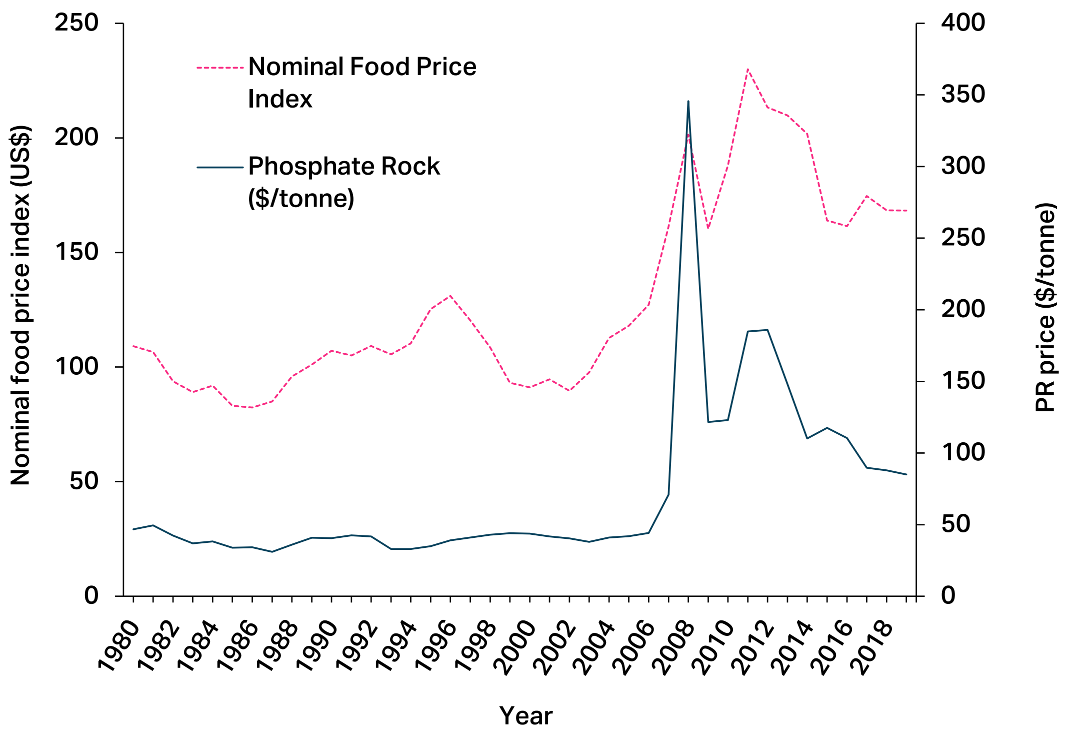


Figure 3.1 FAO food price index (international prices of a basket of food commodities) in nominal terms (data source: FAOstat) and phosphate rock price (US$/t) (Data source: World Bank Commodity Price data) from 1980 to 2019, showing peaks in price between 2007-2008, and again in 2011.

3.1.2 Shifting paradigms of food security

Food security occurs, according to the FAO definition, when "all people, all of the time, have physical, social and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life" (FAO, 2009). This definition is often discussed under a conceptual framework of supply availability and access (including economic access through price), the ability to utilise food (e.g. safe preparation) and stability (i.e. without undue fluctuations in availability and access).

Historically, discussions about food security have focussed on the provision of sufficient food (with emphasis on staple crops supplying calories) for the global poor, particularly in the developing world. More recently this discussion has been nuanced by recognition of two further key issues (Figure 3.2). First, is the need to address the global rise in obesity and the health burden this creates through non-communicable diseases (NCD Risk Factor Collaboration (NCD-RisC), 2016). In many parts of the developed world, obesity is related to inequality and dietary choice and driven by the availability of calorie-dense food that is relatively cheap, whereas nutritionally dense food is expensive (Darmon and Drewnowski, 2015). Poor education around healthy food choices, but perhaps most importantly poor food environments coupled with effective marketing strategies of 'junk food' companies, contribute to this issue. There is an increasing need to focus on providing diets that are healthy and not obesogenic, going beyond providing calories to providing nutrition (Swinburn et al., 2019).

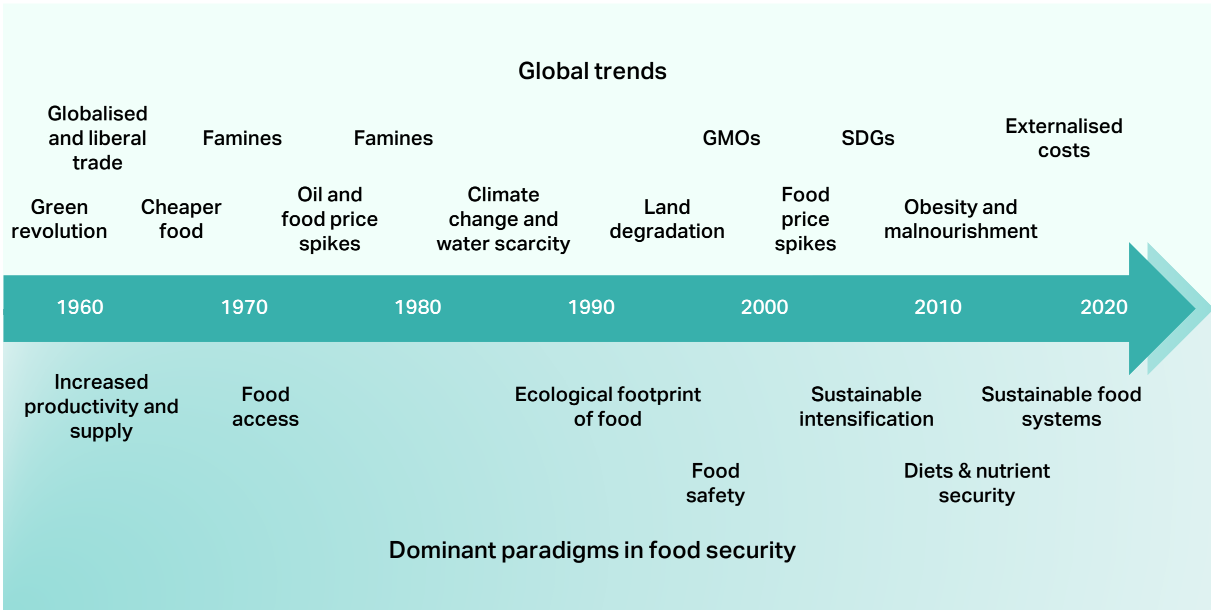


Figure 3.2 Global drivers that have informed discourse on food security since the 1960s to present (GMO – genetically modified organisms; SDGs – sustainable development goals).

Second, is the need to build environmental sustainability into food production systems. Following the 2007–2008 food price spike, and recognising the growing demand for food driven by a rising, more affluent population, projections of demand for food (based on historical trends) suggested a 70–100% demand increase by 2050 (Alexandratos and Bruinsma, 2012). These projections launched a discussion about how to meet that demand by raising the output per unit area (i.e. agricultural intensification) but doing it sustainably. This concept is called sustainable intensification and can be described as a process or system where agricultural yields are increased – or maintained – whilst reducing the environmental impact and without the conversion of additional non-agricultural land (Baulcombe et al., 2009; Benton, 2016; Pretty et al., 2011). Garnett et al. (2013) highlighted that whilst sustainable intensification is an evolving concept, it is only part of what is needed to improve food system sustainability and is not synonymous with food security. Food security has multiple social, ethical and environmental dimensions and to achieve it requires more than just changes in agricultural production. It is also the case that a move to 'sustainable intensification' of food production against rising demand is unlikely to be achieved without increasing environmental impact, albeit at a slower rate than might occur if such approaches were not adopted.

The 2015 Paris Agreement reflected a consensus on the existence of "planetary boundaries", setting limits on global temperature increase related to greenhouse gas (GHG) emissions (IPCC, 2018) and informing discussions on the role of the global food system as a key driver of global climate change, as well as the impact of climate change on food security. This has led to the recognition that "supply-side" measures alone will not deliver sufficient increases in sustainability within food systems (Bajželj et al., 2014; Springmann et al., 2018). For example, redressing the increasing consumption of meat and dairy products in both developed and developing economies may be necessary (Hoekstra, 2012; Leip et al., 2015; Poore and Nemecek, 2018; Röös et al., 2018). In 2021, the term 'Food Systems' was officially adopted at a summit convened by the United Nations to better integrate these issues within the delivery frameworks of the SDGs (United Nations, 2021).

3.1.3 Why the food system needs to change

With more people suffering from hunger and malnutrition than consuming healthy diets (FAO, IFAD, UNICEF, WFP, and WHO, 2017; FAO et al., 2018), the need for transformation of the food system is primarily driven by the need to address malnourishment. This runs alongside the need to reduce the adverse impacts of food production on the global climate system and ecosystem health across all planetary domains. However, transforming the food system to deliver healthy and sustainable diets also addresses P sustainability and climate security (Willett et al., 2019). There is growing recognition that a systemic transformation of the food system is required, and that "business as usual is not an option" (Webb et al., 2020). This recognition comes not just from academic institutions (Fears et al., 2019; Poore and Nemecek, 2018; Ripple et al., 2017) and the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2018) but also from the business community (High Level Forum, 2019; World Economic Forum, 2017). Furthermore, environmental and civil society institutions, including the World Wide Fund for Nature Fund (WWF) (SustainAbility and WWF, 2018) and the World Resources Institute (WRI) (Searchinger et al., 2019) acknowledge the need for change. However, if "business as usual is not an option" what will the future look like for our food systems, and how will this relate to P security?

3.2 Plausible food system scenarios

In the following, we propose a suite of plausible future scenarios for food systems (Benton, 2019). Scenarios are a route to aid decision-making under conditions of high uncertainty (Courtney et al., 1997; Rosa et al., 2017), when past trends cannot necessarily be extrapolated into the future with confidence, and where the future is likely to be shaped by drivers or events which may plausibly lead to very different outcomes. Whilst scenarios may take a variety of forms, a common approach is to identify the two most important drivers which will shape the future, but for which there is great uncertainty about what form they will take (e.g. World Economic Forum, 2017).

A variety of recent exercises have suggested two axes, defining four plausible scenarios (see Benton, 2019 for an overview). These axes are:

Axis 1: Dietary shifts from today's diet, towards food systems that provide healthy food with low-externalised costs to human health and the environment.

The drivers for such a shift include climate change mitigation, health care costs of malnutrition and associated non-communicable diseases, the rise of anti-microbial resistance from intensive livestock production (Rushton, 2015), air quality impacts caused by intensive agriculture and volatilisation of nitrogenous fertiliser (Sutton et al., 2013), the rise in plastic and food waste, and demand to reduce nutrient and pesticide use in agriculture (Chapter 5). Other initiatives that support a shift to healthy and/or sustainable diets include the EAT-Lancet Commission (Willett et al., 2019), Agrimonde foresight work on food security and land use (Paillard et al., 2014), the EU JRC's food systems' foresight study (Bock et al., 2014) and Shared Socio-economic Pathway (SSP) SSP1 for the IPCC (O'Neill et al., 2017).

Axis 2: Change in globalisation towards regional or local food systems.

The dominant view has been that global trade based on economies of scale and comparative advantage is an economic necessity. However, geopolitical trends over the last five years, such as the erosion of the post-WW2 architecture of international cooperation and the rise of inward-looking and protectionist policies driven by increasing global inequality and migration, suggest the future may be shifting towards less globalised trading systems, compared to the trend of the last 70 years. A shortening of supply chains may also be driven by climate change impacts on current food systems and geopolitical instability, incentivising local sourcing to reduce reliance on imported foods. Social change, for example, increasing trust in locally produced foods to support local businesses and communities, might also drive reform (Moberg et al., 2021).

Unforeseen shocks like COVID-19 can also act to increase the resilience of local food systems, or at least highlight the fragility of long supply chains (FAO, 2020). Other exercises also considering the global to local dimension include the EU JRC's food safety foresight study (Mylona et al., 2016), and its scoping study (European Commission, 2013) as well as the IPCC's SSPs (e.g. SSP3 considers more regionalised economies). Both the USA and UK governments publish periodic reports by their intelligence communities, which, in the most recent editions, have highlighted food security scenarios, including the impacts of radical change to the international architecture of trade and cooperation (The National Intelligence Council (USA), 2017; UK Ministry of Defence, 2018). The Millennium Ecosystem Assessment scenarios (Carpenter et al., 2005) also contained a global-to-regional axis. Other discussions around the balance of risks, benefits and costs of trade have included the UK's Climate Change Risk Assessment (Challinor et al., 2018) and the EU JRC's 2030 Foresight Report on Food (Maggio et al., 2015).

Acknowledging the uncertainty associated with these factors, we can define four plausible, alternative, scenarios for food systems, each of which has implications for P, with respect to what food is grown, where it is grown, and how it is grown (Figure 3.3 and Table 3.1).

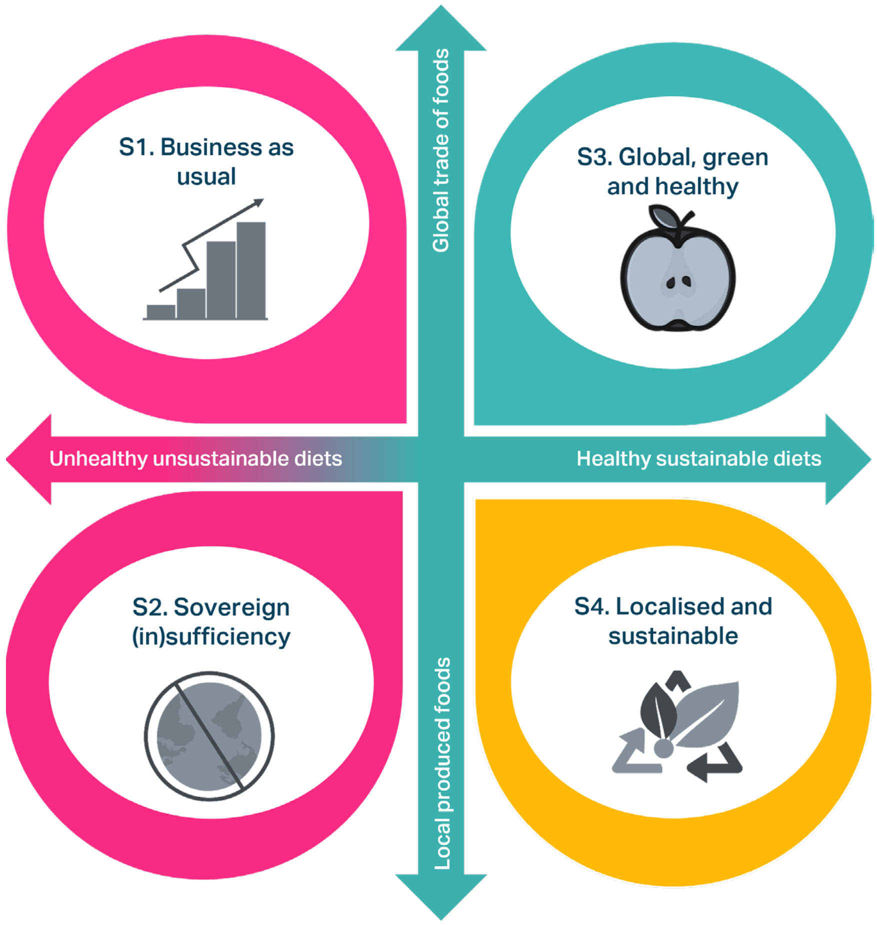


Figure 3.3 Four plausible, alternative, scenarios for food systems, based on axes of global-local connectivity, and degree of dietary shifts. Source: Benton (2019) and World Economic Forum (2017).

Table 3.1 Scenario descriptions of four plausible, alternative, scenarios for food systems, based on axes of global-local connectivity, and degree of dietary shifts Source: Benton (2019) and World Economic Forum (2017).

Scenario	Scenario description
Scenario 1 (S1): Business as usual Unchecked consumption in a globalised world.	Increasing consumption of processed foods based on a few commodity crops. Livestock production continues to increase but is increasingly dependent on grain. Wasting food and over-consuming calorie-dense food continues to be economically rational. Obesity and related ill health and environmental damage increase. "Sustainable intensification" dominates agriculture because meeting demand is the priority. Impacts of the food system exacerbate climate change, nutrient pollution, and biodiversity loss. Land required for climate change mitigation measures (e.g. afforestation, biofuels) adds to land competition. High-tech, intensive cropping systems and intensive livestock production dominate. Economies of scale drive farm amalgamation into bigger units, further marginalising smallholders.
Scenario 1(S2): Sovereign insufficiency Significant reduction in global trade with no change in diets.	Global distrust in international cooperation. Diets are increasingly based on nationally available crops. Diversity of diets shrinks compromising human health, driven by increased reliance on processed food. As the comparative advantage of globally traded food is lost, most countries need to significantly intensify agriculture to become more self-sufficient, leading to increased use of inputs like fertilisers and pesticides. Impacts of the food system exacerbate climate change, nutrient pollution, and biodiversity loss. Endowment-poor, high population countries project more power to ensure food security (e.g. land grabbing). Endowment-poor, low population nations struggle leading to increased human migration. The last two points undermine the national security of endowment-rich countries.

Scenario 3 (S3): Global, green, and healthy Globalised supply chains coupled with a switch towards more "sustainable" and healthy diets	People eat less, aiming for the right amount of calories and with lab meat dominating the alternative protein market. They also produce less waste. Commodity-crop agriculture dominates; nutrition is added through food and agronomic biofortification. Food is processed but with fewer sugar and fats than in Scenario 1. Intensification occurs to meet rising demand but is increasingly focussed in "breadbaskets" of commodity production where there is both scale and intensity of production. "Sustainable intensification" through increasing efficiency is the main focus of environmental concerns. Climate change mitigation actions reduce the need for land-based negative emissions measures, decreasing competition for land. Governments incentivise lower waste through subsidies, food pricing, and waste and food-carbon taxes. Small-scale intensive horticulture increases delivering high-value nutritious crops more widely (especially peri-urban and vertical farming). Adoption of large-scale horticulture by technologically advanced, arid, countries using technological solutions to provide water.
Scenario 4 (S4): Localised and sustainable Circular food systems are diversified to provide healthy diets in a closed system.	Agriculture is local/regional, diverse, with complex rotations, mixed farming, and nutrient recycling. This is necessary to produce nutritionally diverse diets for the population. Without global competition, efficiency should be built into food systems, because local systems struggle to produce an excess of food. People eat less, aiming for a healthy diet that is sustainably produced. Rather than eat hyper-processed foods, people switch to whole foods cooked at home because food is more expensive. Agricultural policy is driven by nutritional needs and environmental protection, not just economic growth. Health costs are avoided because people eat healthily, and, along with circularity, agriculture is more diversified and landscapes more heterogeneous. Nutrient losses are reduced leading to environmental recovery and a reduction in greenhouse gas emissions from land-based food systems alleviates impacts on climate change. Localised systems will exacerbate among-country inequality, which may lead to aggressive land-grabbing (as with Scenario 2). Localised food systems experience low resilience to local extreme weather events, but are not exposed to risks from interrupted trade (which may become more common as climate impacts increase). Food systems reflect local conditions, creating diet seasonality and locally adapted produce.

The scenarios describe plausible future food systems and define different research and policy agendas. We stress that scenarios do not describe the "most likely future"; their main value lies in providing stress tests to aid in future policy development. However, these scenarios do suggest the future farming system – and the associated needs for, and impacts on, P – may not reflect a linear extrapolation of current farming systems. Therefore, farming systems should avoid being locked into an undesirable future (e.g. creating a greater reliance on fewer crops when the future may require diversification), but instead, be designed to be adaptive. Table 3.2 (presented at the start of the Solutions section in this chapter) identifies the implications for P, of sustainable transformations of the food system.

3.2.1 Implications for farming and the food system of a transition to healthy diets

Currently, about 2.5 times more cereals are grown worldwide than is needed to meet US dietary guidelines and only a fifth of fruit and vegetables needed are grown (KC et al., 2018). The EAT-Lancet report (2019) states that transformation to healthy diets by 2050 will require the average global consumption of fruits, vegetables, nuts and legumes to double, and consumption of foods such as red meat and sugar to be reduced by more than 50%. Currently, there is a stark global misalignment between what we eat and what we need for a healthy diet (EAT, 2019), though with regional variations (Figure 3.5 - see the following page spread). A move to a diet that supports a preventative health care system, therefore, has significant implications for agriculture, and consequently phosphorus.

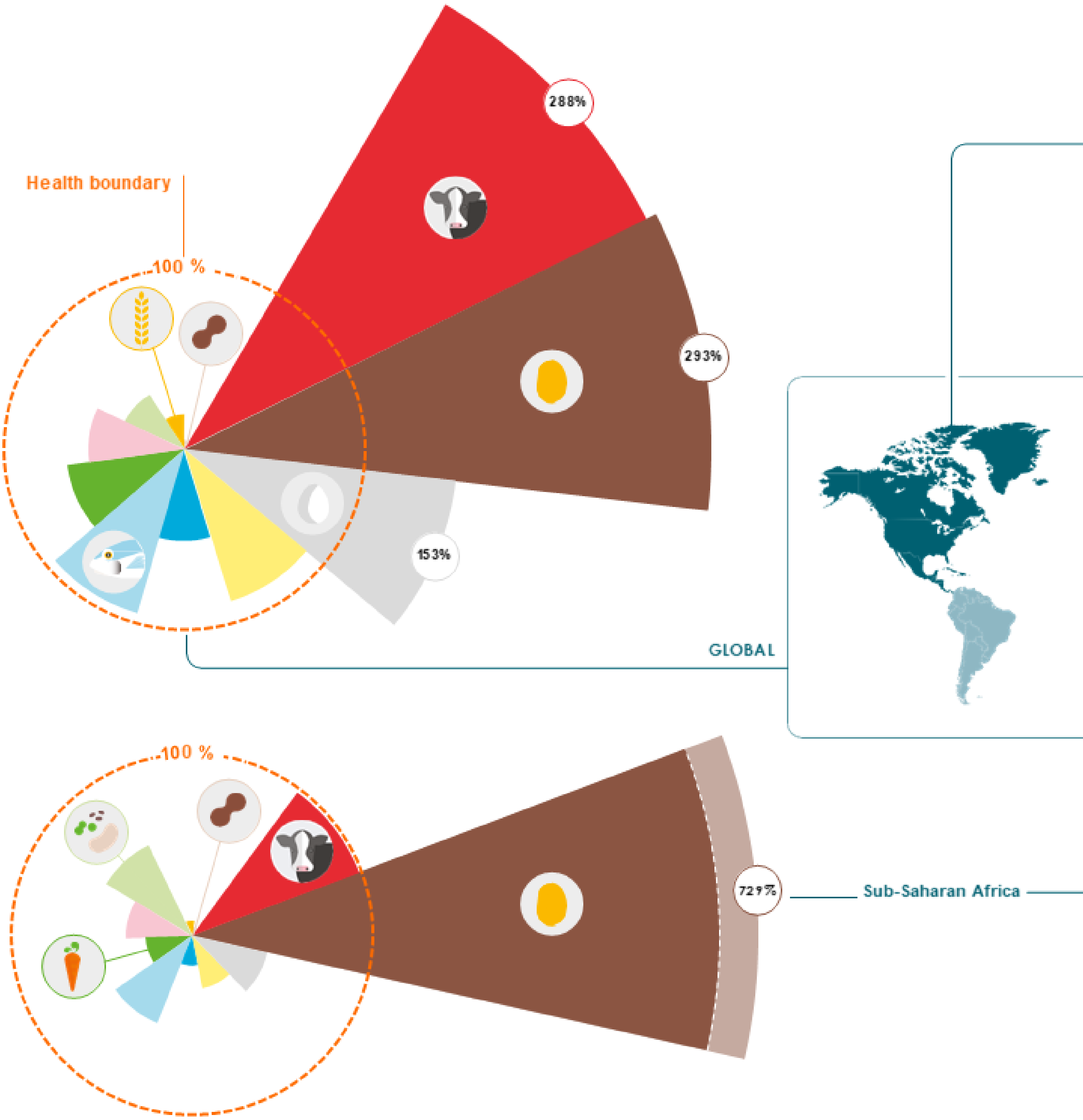
A sustainable food system implies more flexitarian diets (Machovina et al., 2015; Springmann et al., 2018). Globally, eating less meat, especially red meat, and animal products, is required to reduce the emission of GHG from the global food system, a significant step in mitigating climate change (IPCC, 2018). Life cycle analysis used to underpin this shift is based on CO_2 equivalent emissions per unit mass of meat (de Vries et al., 2015; McAuliffe et al., 2018a). However, replacing the 'mass' with 'nutrient content' of meat as the functional unit can dramatically alter relative emissions intensities associated with different livestock systems. In some cases, cattle systems can outperform pig and poultry systems (McAuliffe et al., 2018b).

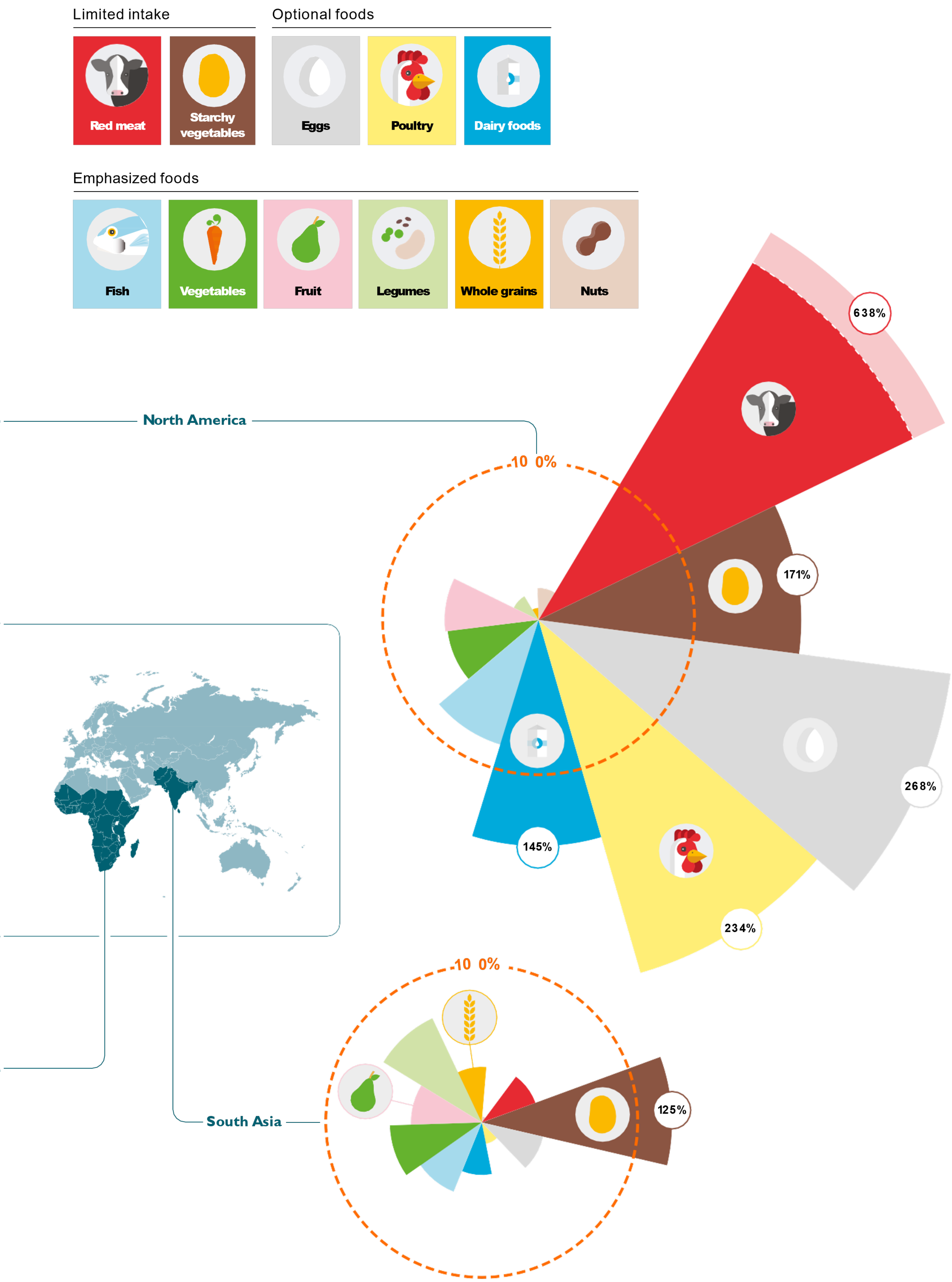
Animal welfare can, and already does, influence societal dietary behaviours, which can have global impacts on agricultural systems, tending to drive a decrease in intensive livestock production and support more local food production with higher priorities for animal welfare. However, it is not clear if a reduction in meat consumption will lead to a compensatory increase in plant-based protein consumption. On average, globally, people consume too much animal protein (Figure 3.5), and protein in general (WRI, 2016). Thus, there may be little nutritional need to substitute the same quantity of animal protein with plant-based protein. A shift to more plant-based diets creates new challenges and opportunities for meeting the fresh fruit and vegetable demands of urban populations. The substitution of animal protein with insect protein in human diets warrants further investigation (Durst et al., 2010; Martin, 2014) (Figure 3.4), as does the use of insect protein as an alternative or supplementary source of animal feed (Leiber et al., 2015). What is clear is that our food systems are unpredictable and transforming, highlighting a significant challenge for future P management.

In the following section, we discuss the key challenges and solutions with respect to achieving more sustainable P use to deliver greater food security.

Figure 3.4 An 'Essento Insect Burger' contains 30% organic mealworms (Tenebrio molitor) farmed in Switzerland, along with chickpeas, bulgur, spelt, carrots, celery and a spice mix. The burgers, made by Swiss startup, Essento, are available (as of August 2021) in all larger Coop branches in Switzerland and for food service companies in Austria, Germany and Switzerland. Photograph copyright of Essento.

Figure 3.5 What we eat versus what we need for a healthy diet as the global average (left middle) and in North America, South Asia and Sub-Saharan Africa. A healthy diet is indicated by the dotted line orange circles (the health boundary). Food types that expand beyond the health boundary are eaten in excess, whilst more should be consumed of the food types that fall short of the health boundary. The figure shows that globally we consume far more red meat and starchy vegetables than required for a healthy diet, and far too few legumes, whole grains, fruit and vegetables. Source: adapted from (EAT, 2019). Reprinted from the EAT-Lancet Commission Summary Report, with permission from Elsevier.




Limited intake
Red meat
Starchy vegetables
Optional foods
Eggs
Poultry
Dairy foods
Emphasized foods
Fish
Vegetables
Fruit
Legumes
Whole grains
Nuts
North America
100%
638%
171%
268%
234%
145%
South Asia
100%
125%

3.3 Challenges

Challenge 3.1: Business as usual is unsustainable: we must produce healthier foods, using appropriate phosphorus inputs

Our food system is a significant cause of nutrient pollution in terrestrial, freshwater and marine ecosystems, and of global climate change, while more than half the global population are acutely hungry, malnourished, overweight, or obese. The public health and ecological costs of the current food system exceeds the economic value of agriculture. Systemic transformation is required for food systems to become environmentally sustainable and provide nutritional security for all. Sustainable phosphorus strategies must directly support—not hinder—this transformation. On the current path, the global food system will increase the mining of finite phosphate rock to produce fertiliser, feed additives and food supplements, and is not tracking towards a circular phosphorus system (driven on recycled phosphorus inputs).

A systemic transformation of the food system is required. Under a 'business as usual' scenario, including current inefficient food production and consumption practices, it has been estimated that a 70-100% increase in crop production will be required to feed an expected global population of 9.7 billion in 2050 (based on 2005 levels) (Tilman et al., 2011). However, these estimates misinterpret underlying projections and ignore recent and potential production gains across the whole food value chain. Hunter et al., (2017) argue that an increase in production by 25-70% will be sufficient although simply intensifying current food systems to meet this demand is not sustainable. The current food system, including societal dietary choices, is widely acknowledged as eroding human and planetary health (IPCC, 2019; Willett et al., 2019).

Dietary-related malnourishment (including both under-nutrition and over-consumption of calories) is the prime global determinant of morbidity and is affecting every society in every country (Development Initiatives, 2018; GBD 2017 Risk Factors Collaborators, 2018). In 2016, 821 million people were acutely hungry, with an additional 2 billion undernourished. At the same time 2.3 billion people were overweight or obese (FAO, IFAD, UNICEF, WFP, and WHO, 2017; FAO et al., 2018). There are now more obese adults in the world than underweight and, if current trends continue, by 2025 there will be more severely obese adult females than underweight (NCD Risk Factor Collaboration (NCD-RisC), 2016). Simply put, out of every nine people, one person is starving, three people are undernourished, and three people are overweight or obese. In the developed world, at least, obesity is often linked to poor diets and poverty (Darmon and Drewnowski, 2015), because many cheaper foods are rich in calories but poor in nutrition. These foods are based on major commodity crops providing oil, sugar and starch (Development Initiatives, 2018). The ill-health burden that arises from malnourishment is also a growing economic burden (FAO, IFAD, UNICEF, WFP, and WHO, 2017; FAO et al., 2018). When combined with the environmental costs of producing foods (Burlingame et al., 2012; Kahiluoto et al., 2014; SustainAbility and WWF, 2018; West et al., 2014), including impacts of agricultural nutrient use, the cost of the current food system far exceeds its current economic value (Collins et al., 2018; Fitzpatrick et al., 2017; TEEB, 2018).

Food production practices must change to avoid further P-related environmental damage. For example, there is a need to reduce the inputs of non-renewable PR to produce food through increased nutrient recycling and more efficient utilisation of P in agricultural systems. The use of mineral fertilisers has played a crucial role in feeding billions of people over the past 60 years. Mining PR for use as fertiliser, and other human changes to the global P cycle, have increased the rate of P movement from mineral rock deposits to the ocean by four-fold (Falkowski et al., 2000; Smil, 2000). Phosphorus losses from land to surface waters have increased globally from around 5 to 9 Mt P year^{-1} over the 20th century (Beusen et al., 2016). Significant P losses are delivered from food production systems, such as livestock wastes or runoff from P-rich agricultural soils, or as excreted wastes, in the form of sewage discharges, into waterbodies with detrimental impacts on ecosystem and human health (see Chapter 5). Low P use efficiency (PUE) throughout food systems (from farm to fork and beyond) is widely acknowledged in Europe (Fischer et al., 2017; Muhammed et al., 2018; van Dijk et al., 2016), China (Lou et al., 2015), Brazil (Fischer et al., 2018; Withers et al., 2018) and the USA (Suh and Yee, 2011) (see Chapter 4). Low PUE drives losses that could be recouped to drive more sustainable food systems in these regions and others. More widespread adoption of strategies to increase PUE is urgently needed.

Challenge 3.2: Increasing global consumption of animal products is increasing phosphorus demand

The amount of phosphorus required to produce the average per capita global diet has increased by 38% in the last 50 years, due to the rise in consumption of animal products, increase in average per capita consumption and increased food waste. Excluding phosphorus-efficient grass-based systems, a large proportion of cropland is needed to support intensive meat and dairy production through concentrated animal feeding operations. This trend is driving increased mining of phosphate rock for fertilisers, animal feed and supplements. Unhealthy diets, including overconsumption of animal products, are also a significant contributor to non-communicable diseases.

Global consumption of animal products is rising, with a significant increase in countries such as Brazil and China, although at levels below those in industrialised countries (Westhoek et al., 2015).

The amount of P required to produce the average per capita global diet has increased by 38% in the last 50 years (Metson et al., 2012), due to the rise in consumption of animal products and increased food waste (and calorific intake in some regions). The current average per capita protein intake in the EU is about 70% higher than recommended (WHO, 2003). In Europe, ~20% of total P imported is for animal feed, whilst 60% of P in harvested crops is used in animal feed, and only ~30% is directly consumed by humans in plant-based food (van Dijk et al., 2016). In addition to grass-based systems, a large proportion of cropland is needed to support intensive meat production in concentrated animal feeding operations. Livestock density is, therefore, a major driver of the overall low PUE in national and regional food systems, both in grazed and in animal feed supplemented systems (e.g. Withers et al., 2020; Rothwell et al., 2020).

Global demand for P is highly influenced by diets that are high in animal protein (especially red meat) (Metson et al., 2012). For example, up to 16 times more P (and other resources) is required to produce a unit of beef from a concentrated animal feeding operation than to produce plant-based proteins (Metson et al., 2012). However, changing eating habits at the population scale is complex because behaviours are based on a large set of factors, unique to each person and their environment (see Chapter 8). Further, a third of all food produced globally is wasted (FAO, 2011), which contains P, resulting in a significant environmental and economic burden. For P, this means a significant unnecessary P demand and associated nutrient pollution (Kummu et al., 2012). We are currently on track for a more wasteful future scenario (i.e. 'unchecked consumption and sovereign (in) sufficiency') in which P demand will increase.

Challenge 3.3: Balancing intensive agriculture with low input farming

Agricultural intensification increases productivity yet increasing phosphorus inputs to crops can also over-enrich adjacent land and waterbodies with nutrients. Lowering phosphorus inputs reduces environmental risk and promotes biodiversity but may restrict yield in the long-term. Strategies need to provide the right balance of intensification to avoid the need to convert more land to agriculture. Optimising the multitude of costs and benefits and taking account of direct and indirect impacts can be challenging and context specific. The challenge we face is in developing low phosphorus input farming systems which can sustain food production.

Agricultural intensification using mineral P fertilisers, high-yielding crops, irrigation, and pesticides has contributed significantly to the large increases in food production over the past 60 years. Nevertheless, shortages of land that can be converted to agricultural use without inflicting yet greater damage on the environment (Lambin et al., 2013) mean there is still a need to increase the output of food, and other products like fibres, on existing agricultural land, unless there are dietary shifts that change demand. A similar scenario is emerging for aquaculture.

However, such intensification has long been known to alter the biological interactions and patterns of resource availability in ecosystems, with serious local, regional, and global environmental consequences (Matson et al., 1997). Some intensification of food production may well be necessary for sustainable human development, however, the increase in intensive agriculture has complex impacts on P demand and losses to waters (Ockenden et al., 2017). For example, while high-yield farming might increase PUE (Syers et al., 2008), increasing P use per hectare also increases the accumulation of unused P in the landscape both on land and in waters (Powers et al., 2016; Withers et al., 2014a). Historically in the global north, farmers have been advised to over-supply P to avoid risking yield loss (so-called insurance-based farming; Withers et al., 2014b), and areas with intensive livestock systems add further manure P loading pressures on the landscape, greatly exacerbating pollution risks to neighbouring water bodies (Leip et al., 2015; Powers et al., 2016; Withers et al., 2014b).

In some regions, such as the tropics, the need to increase food production is driving farming on 'P fixing' soils. Phosphorus readily binds to these soils, meaning a lower proportion of P applied as fertiliser is available for plants (Batjes, 2011; Sanchez et al., 2003). To overcome this, farmers often over-apply P well above plant requirements, increasing, over time, the risk of P transport to water bodies in runoff (Withers et al., 2018). Roy et al. (2016) estimate that intensification of the 8–12% of global croplands overlying P-fixing soils in 2005 would require 1–4 Mt P $year^{-1}$ to overcome P fixation limits, equivalent to 8–25% of global inorganic P fertiliser application that year. This imposed P 'tax' is in addition to P added to soils and subsequently harvested in crops, and is projected to double to 2–7 Mt P $year^{-1}$ for scenarios of cropland extent in 2050.

Continued high P inputs inevitably lead to high P losses from food systems (Doody et al., 2016; Withers et al., 2020). There is little evidence to date that this relationship can be effectively decoupled by best land management practices without transforming food systems. Nevertheless, PUE can be increased in some regional food systems by a range of measures. These include: reducing unnecessary P use in fertilisers and feeds; careful management of the P fertilisers and manures that are applied; breeding more P-efficient crop cultivars and novel and more precise agronomic practices that reduce P input requirements and better utilisation of both applied P and existing legacy P reserves in soils (see Chapter 4).

Currently, efforts to combat food insecurity in Sub-Saharan Africa (SSA) focus on agricultural intensification, although an alternative approach emphasising diets, health and the environment has been put forward (Global Panel on Agriculture and Food Systems for Nutrition, 2020). Given the high soil nutrient depletion in this region, replenishing soil fertility is a major component of such efforts (Nziguheba, 2007; Nziguheba et al., 2016). The dominant 'productivist' mentality that still governs the food system, means that farmers and policymakers will continue to favour measures that maintain or enhance production. There remains a reluctance to implement measures that might, for example, take land out of production (Inman et al., 2018) or reduce animal density (Metson et al., 2012; Withers et al., 2020).

Challenge 3.4: Many farmers lack access to phosphorus, threatening their livelihoods

Currently, 1 in 7 farmers cannot access or afford phosphorus fertilisers to increase productivity, reducing their ability to maintain food security and livelihoods. Those farmers most affected are rural smallholder farming families, particularly in less economically developed countries, but also in some more economically developed countries. There are marked global inequalities in access to phosphorus as a resource, leading to substantial inequalities in the distribution of risks to food security.

We currently produce enough food to feed 10 billion people; about 30% more than the global population. As Holt-Giménez et al. (2012) point out, hunger is caused by poverty and inequality, not food scarcity, at a global scale. Those that live on less than US$2 a day, mainly subsistence farming families, cannot afford to buy sufficient food. If the priority is better health for all, global food systems must stop fuelling diets with adverse public health impacts, requiring a systemic change to global food production systems (Global Panel on Agriculture and Food Systems for Nutrition, 2020). A significant proportion of industrially-produced grain crops goes to biofuels and confined animal feedlots rather than food for the 1 billion hungry (Holt-Giménez et al., 2012). Therefore, calls to double food production by 2050 only apply if we continue to prioritise the growing population of livestock and automobiles over hungry people.

Most of the world's food insecure are marginalised families in urban and rural environments. The latter are also typically smallholder farmers (Dixon et al., 2001), who struggle to access fertilisers. According to the FAO (2014), there are more than 500 million family farms globally, producing 80% of the world's food, although Ricciardi et al. (2018) suggest this figure is closer to 35%. In many low-income countries, including in SSA, South Asia, East Asia and the Pacific (excluding China), around 70–80% of farms are smallholder farms (Lowder et al., 2016).

The livelihoods of more than 2 billion people depend on smallholder farms (Lowder et al., 2016). Yet many of these smallholder farmers (particularly in low-income countries) lack sufficient access to P fertiliser markets due to poverty, low purchasing power or because they lack access to credit (Dixon et al., 2001; Druilhe and Barreiro-Hurlé, 2012; IATP, 2005; McIntyre et al., 2009; Runge et al., 2003). Farmers need access to P to replace the P removed in harvested crops and other losses and maintain fertile soils for crop growth. Affordable and sustainable access to P fertilisers is therefore imperative to ensure food security at a national scale, and the food and livelihood security of smallholder and marginal farmers (Weber et al., 2014).

Fertiliser prices are increasing in the long-term and may be subject to further short-term price spikes, (Mew, 2016). In 2008, PR prices spiked by 800%. The cause of this was complex and discussed earlier (see also Chapter 2). Elevated fertiliser prices made farmers more prudent with their fertiliser use and eventually led to a crash in demand for phosphorus. After this price peak, phosphate rock price dropped significantly, but is on average more than two times higher than before the price peak (Mew, 2016) (see Chapter 2).

The use of fertiliser subsidies has been widespread in SSA for decades and is highly controversial. For example, in Nigeria between 1980 and 2010, it has been claimed up to 90% of subsidised fertiliser was bought by officials and sold to private companies (Propcom Mai-karfi, 2014). Only 11% of farmers received subsidised fertilisers, which were often adulterated, damaged, and delayed (Banful et al., 2010; Udo, 2013). Fertiliser subsidies in Africa are politically popular due to their immediacy and visibility (Druilhe and Barreiro-Hurlé, 2012), however, work is needed to ensure they deliver on their aims, and importantly are not impacted by corruption. The African Union during its 30th Assembly of Heads of State and Government in January 2018, declared 2018 as the African Anti-Corruption Year, aiming to ensure better cooperation and mutual legal assistance, and secure stronger international cooperation in dealing with corruption (African Union, 2018).

In some low-income countries, insufficient use of fertilisers and soil erosion has led to substantial nutrient depletion of soils, constraining agricultural productivity. The most vulnerable are subsistence farmers, many of whom are already seeing production levels fall as soil fertility declines, such as African farmers practising shifting cultivation or cultivating marginal lands (Nziguheba et al., 2016), and Brazilian livestock farmers relying on the degrading pastures in the Cerrado region (Pereira et al., 2018). In Africa, fertilisers cost 2 to 6 times more than in Europe, the USA or Asia (Chemonics and IFDC, 2007; Sanchez, 2002), predominantly due to poor infrastructure which can create high costs for over-land transporting, stocking, and distribution (Cordell et al., 2015; Druilhe and Barreiro-Hurlé, 2012). These high costs can undercut the trade competitiveness of agricultural produce (Keyser and Tchale, 2010). African agriculture has enormous potential to drive equitable and sustainable economic growth across the continent, but a keystone in its success will be access to and sustainable management of P, and other nutrients (Chianu et al., 2012). Without change, it has been predicted that insufficient P inputs to African soils could lead to a further 30% reduction in crop yield by 2050 (van der Velde et al., 2014). Of course, the expansion of agriculture in any region of the world should be balanced against the adverse impacts of all agriculture on biodiversity. Greater yields per hectare are desirable in this context, but the conversion of biodiverse habitat to farmland may not be (Benton et al., 2021).

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3.4 Solutions

Solution 3.1: Managing phosphorus sustainably can support a shift to healthier diets

Global food systems must produce, actively support, and provide access to nutritious food and diets for all. This shift, from 'market-led' to 'sustainable' food security, can reduce phosphorus demand and adverse impacts on ecosystems and society. Concurrently, strategies to deliver better phosphorus sustainability, including circular phosphorus value chains, can benefit agricultural economies, whilst effective monitoring systems, data sharing, and knowledge exchange can ensure strategies adapt to a transforming food system.

Transforming the food system to deliver healthy diets within planetary boundaries implies lower production of red meat and the 'Big Three' grains of wheat, rice and maize (corn), and increasing the diversity of grains (e.g. pulses and lentils) and producing more fruit and vegetables (EAT, 2019). Such a shift may dramatically change fertiliser requirements. Whilst reductions in livestock numbers would significantly reduce global demand for P fertiliser to grow feed (Metson et al., 2012) and increase overall food system P efficiency (Withers et al., 2020), an increase in the production of fruit and vegetables could require more P fertiliser. It could also require more water either from rainfall or through irrigation, which will increase the risk of P loss from farmed soils to waters. Indeed, any change to quantity and types of foods produced will impact mineral P application rates, formulations and timing, the efficiency of P use, and P mobilisation and export within fields, and thus influence P loads entering wastewater treatment works (Forber et al., 2021) and waterbodies (see Chapter 5). Where foods are grown will also affect P demand and the environmental footprint due to different soils, climates, cropping systems and knowledge systems, especially if there is a shift to local and regional food systems. Climate change impacts P input requirements, agricultural output and subsequent losses to water, adding yet another layer of complexity that will vary considerably between different regions (Forber et al., 2018).

Strategies and measures to improve sustainable P use can support the transformation towards more sustainable food systems (Withers et al., 2015). For example, improving PUE can reduce farmers' fertiliser input costs, while reducing their reliance on mineral P fertiliser by increasing access to local recycled P markets, including animal manures and slurries produced on local livestock farms. This, in turn, can create new business opportunities for nutrient recovery in a circular economy, where wastes and residues become products in their own right. Furthermore, providing better information on which farming systems best match P availability in local wastes can support spatial planning and decision-making to optimise agricultural productivity. For example, coupling livestock and arable food production systems to support nutrient reuse between them, and discouraging or providing better options for farming on P fixing soils represent two relevant opportunities to support this transition.

Strategies to deliver sustainable P management must recognise co-benefits and evolve alongside a transforming food system (Table 3.2). The use of 'dynamic adaptive policymaking' (Haasnoot et al., 2013; Walker et al., 2013) supported by effective monitoring systems (e.g. to assess nutrient concentrations in water bodies), data sharing, and effective communication of scientific evidence to both the public and policymakers (Brownlie et al., 2017) can help in this regard.

Table 3.2 Implications for phosphorus (P) of transforming the food system

Sustainable pathways for transforming the food system	Implications and co-benefits for phosphorus
Pathway 1: Produce appropriate food for nutritious diets Systemic changes to the food system are required that address the current disconnect between what we produce (e.g. red meat and the "Big Three" grains: wheat, rice, maize (corn)) versus a recommended balanced diet (Figure 3.4; EAT, 2019). We need to grow more vegetables, fruits, legumes globally.	Producing different foods can change the P fertiliser demand associated with different crop types. For example, shifting from cereal crops to legumes may reduce P fertiliser requirements because the latter is more P-efficient to produce (Lyu et al., 2016). This may also require changes in fertiliser formulations for different crops and geo-climates.
Pathway 2: A shift away from diets with adverse public health impacts Changing diets is one of the single biggest food transformation levers, especially in high meat-consuming countries (Ranganathan et al., 2016). Unhealthy diets (including red meat consumption) are one of the greatest risk factors to human health (e.g. cardiovascular disease, obesity) (GBD 2016 Risk Factors Collaborators, 2017). The triple burden of food insecurity means that 2 billion people are obese or overweight (increasing in every country, including low-income countries); 2 billion people have micronutrient deficiency (e.g. iron, vitamin A); 816 million people are hungry (exacerbated by climate change & conflict, even in high-income countries like Australia where 10-20% are hungry) (FSIN 2018, 2019).	A shift towards healthier plant-based diets globally will result in a lower overall P footprint of the food system, delivering global-scale gains in PUE. For example, the average person in the high-meat consuming nations of Argentina and the USA has a P footprint of over 6 kg P $year^{-1}$, compared with those in India (~1 kg P $year^{-1}$). This is predominantly a result of per capita meat consumption, not total food consumption (Metson et al., 2012). Knock-on effects of dietary change on increased P loading to wastewater treatment centres may provide an additional concentrated source of secondary P for reuse if recovered effectively (Forber et al., 2021), or greater adverse impacts on aquatic ecosystems, if not recovered before the discharge of effluent to waters.

Sustainable pathways for transforming the food system	Implications and co-benefits for phosphorus
Pathway 3: Decreased environmental footprints of food production and consumption The cost of the environmental and health burden of the current food system far exceeds the value of global agriculture (Collins et al., 2016; Zhang et al., 2017). Climate change, water scarcity and pollution, energy scarcity and pollution, obesity and the non-communicable food-related disease epidemic (diabetes and cardiovascular health, cancers) means business as usual commodity-crop based agriculture is not an option.	Measures to reduce the environmental footprints of food production often have a lower P footprint (Metson et al., 2014). For example, millet is not only a nutritionally dense grain (high in calcium and iron) but has a low carbon footprint and low pesticide and fertiliser requirement (ICRISAT, 2018). Reducing P losses from agriculture can reduce water pollution by minimising fertiliser losses in eroded topsoil, surface entrainment of P-rich manures and slurries, and the flushing of desorbed P through P-saturated soils and groundwaters to rivers, lakes, estuaries and the coastal zone.
Pathway 4: Reduce food waste in the supply chain Pre- and post-harvest food waste globally are estimated at 30-50% (varying widely across value chains and regions). This results in waste of embodied energy and resources (e.g. nutrients) used to produce, process and transport the food, and occupation of valuable space in landfills from which methane is released upon decomposition, in addition to the cost to consumers and food producers (FAO, 2011).	Reducing food waste directly reduces P wastage because, like all organic waste, food processing waste and food waste contains P and has a P footprint associated with its production. Currently, 80% of input P is wasted along the whole P value chain (Cordell et al., 2009), half of which could be post-harvest. Reducing this loss would reduce global demand for mined non-renewable phosphate and/or make P more available for reuse such as the use of compost or digestate.

Sustainable pathways for transforming the food system	Implications and co-benefits for phosphorus
Pathway 5: Shorten food value chains (where appropriate) Producing food closer to where it is consumed can increase food security in the face of shocks like fuel shortages or floods that affect transport routes. Shorter supply chains reduce energy, waste, middle-management, transport and cooling, to deliver the same unit of food.	This pathway presents a potential trade-off for phosphorus. There is a risk that local food systems can be potentially less P efficient at the farm scale, depending on local soils, technical and knowledge systems. However, at the larger system scale, this would reduce P wastage in the post-harvest value chain by, for example, reducing the P embodied in food commodities that end up as food waste due to spoilage or market excess.
Pathway 6: Increase food access Currently, over 800 million people lack sufficient access to food. There is a need to increase financial access, physical access, and food literacy, to in turn improve the health, productivity, quality of life and livelihoods of this food insecure group.	Most of the world's food insecure are rural smallholder farming families – the same group who struggle to access fertilisers. Around 1 in every 7 farmers cannot access fertiliser markets (McIntyre et al., 2009). Fertiliser prices are increasing and may be subject to further price spikes. Increasing access to fertilisers or local nutrients (or access to loans/credit) can increase crop yields and hence income and food security for farming families.
Pathway 7: Consider the food system's whole value chain, beyond agriculture The FAO and the Organisation for Economic Co-operation and Development (OECD) have adopted the term 'food systems' to acknowledge and assess the complex links between consumption patterns, food processing and retail value chains, farmer livelihoods, public health, environment and agricultural inputs. Adoption of this consistent framework will provide insights into opportunities for addressing unsustainable practices across currently obscured value chains.	In addition to on-farm P management, such as the timing of fertiliser applications to crops and grass and the management of manures and slurries, many opportunities occur before or after the farm, like recycling P in organic waste (or reducing losses), fertiliser production, food storage, processing, and retail. In addition, shifting consumer preferences can have a significant impact up the value chain, in terms of how sustainably crops are grown (including their P efficiency), and which foods are produced (such as animal or plant-based proteins).

Solution 3.2: Shift global consumption of animal products towards plant-based diets

Reduced consumption of animal products especially from intensive production systems in some regions may reduce global agricultural phosphorus demand and contribute to healthier environments. Increased awareness amongst policymakers and the public of the environmental impacts of phosphorus use in food production, and the human health risks of excessive consumption of animal products, will be an essential driver of change. Knowledge exchange between academics, stakeholders and the public can help identify solutions to support a transition to more phosphorus sustainable consumer behaviour, as could policy and regulatory changes (including internalising the environmental costs into food pricing).

Reduced consumption of animal products from concentrated feeding operations in some regions will significantly reduce global agricultural P demand (Elser, 2012; Ma et al., 2012; Metson et al., 2012; Schröder et al., 2011; Suh and Yee, 2011). Reducing the consumption of animal products (for those people eating excess amounts) will contribute to healthier humans and environments (EAT, 2019; Elser and Bennett, 2011), especially for high-meat consuming countries. Plant-based diets typically have a lower P footprint; 1 kg of P fertiliser can produce over 3000 kg of starchy roots or 16 kg of beef (Metson et al., 2012). Hence, if diets shift to plant-based, this could reduce global P demand by some 50%. Whilst this may not be realistic in the short term (e.g. 5 years), even modest shifts to plant-based diets will have a significant impact on P demand (Metson et al., 2012). However, such a shift can increase P loadings to wastewater treatment works instead of agriculture, although if effectively recovered this could provide an additional source of secondary P for reuse in agricultural systems (Fober et al., 2021). If this P is not recovered, however, P availability and loading impacts on aquatic ecosystems could increase. Therefore, a shift to more plant-based diets will require more investment in P recovery technologies and wastewater treatment infrastructure in parallel, if off-site P impacts are to be managed.

Changing eating behaviours is possible (IPCC, 2019; Loken and DeClerck, 2020; Ranganathan et al., 2016). Improving public and policy awareness of the impacts of P use in food production will help to change societal and policy support for sustainable P practices. Currently, it would be reasonable to presume most people do not buy foods based on the P impacts of their production. A combination of factors influences each consumer's buying choices, such as cost, convenience, availability, taste, appearance, positioning (e.g. at eye level), marketing health, environmental impacts (e.g. impacts on climate change, biodiversity), animal welfare, and buying habits. It is, therefore, important to ensure that products with low P footprints are aligned with these criteria, so consumers are more likely to purchase them (see Chapter 8). Knowledge exchange between social scientists, stakeholders and the public will help identify solutions to support a transition to more P sustainable consumer behaviour. Efforts are increasing in this area. For example, networks and platforms have been developed at the national and international scales to support knowledge exchange (e.g. the European Sustainable Phosphorus Platform, the Sustainable Phosphorus Alliance, and the Global Phosphorus Research Initiative). These bodies support networks across multiple sectors providing evidence on sustainable P issues across different scales. The combined efforts of these groups are powerful, and they offer an essential conduit through which emerging approaches and evidence can be effectively integrated across scales to support the transition of food systems towards greater P security. For example, recent developments across multiple fields in P sustainability include bringing together international experts to identify national-scale improvements (Macintosh et al., 2019), conducting stakeholder analyses to support coordination across sectors (Lyon et al., 2020), identifying barriers and solutions to fostering pro-environmental behaviour (Okumah et al., 2020), and delivering cost-benefit analyses including impacts on human health associated with an increase in P recycling (Tonini et al., 2019).

Solution 3.3: Integrated landscape strategies to improve phosphorus use efficiency and reduce losses

There is an opportunity to develop novel land-use planning approaches to support more sustainable phosphorus use across multiple and interacting contexts. These include agricultural production, ecosystem and human health, local economies and regional capacity for institutional planning and coordination. Sustainable farming systems in which animal and crop production are more integrated and animal residues and manures are treated as valuable phosphorus resources, will support efforts to increase phosphorus use efficiency within landscapes while reducing negative impacts on aquatic and terrestrial ecosystems.

There is a recognised need for integrated land-use planning that considers how to balance multiple needs including resource stocks and flows, energy dynamics, flood retention, urban regeneration, biodiversity, and climate change mitigation (Estrada-Carmona et al., 2014; Macintosh et al., 2019). Robust data on soil P content and the amount of P applied to soils in fertilisers and manures and other residual flows will support more effective P management in sustainable agricultural intensification (Macintosh et al., 2019). Methods to mine 'legacy P' (i.e. P stored in soil that is not immediately available for plant uptake), such as through P efficient cultivars, root management, rhizosphere microbiome engineering and rhizosphere interactions (Lemaire et al., 2021; Rowe et al., 2016; Schneider et al., 2019), can be used in combination with careful management to improve PUE throughout the food production chain, from mine to fork (see Chapter 4).

Field data suggest that sustainable agricultural intensification can protect biodiversity by boosting yields on existing farmland and sparing undisturbed habitats from being brought into production (Balmford et al., 2018; Garnett et al., 2013). Whilst high yield farming generates more externalities per unit area (e.g. greenhouse gas emissions and nutrient losses), Balmford et al. (2018) argue these metrics underestimate the impacts of low yield farming. In some areas, increases in yield will be compatible with environmental improvements. In others, yield reductions or land reallocation will be necessary to ensure sustainability and to deliver other desirable benefits (e.g. wildlife conservation, carbon storage, flood protection, and recreation). For example, the opportunity exists to better integrate nature-based solutions within agricultural catchments to deliver both reduced greenhouse gas emissions and more sustainable P use through the food system (Seddon et al., 2019). It is important to recognise that urbanisation and food system efficiency drive important feedbacks with climate systems. An overall increase in food production does not mean yields must increase everywhere or at any cost. The challenge is context- and location-specific and requires careful land-use planning and reform (Balmford et al., 2018) based on sound scientific evidence to support decision-making (Garnett et al., 2013).

Solution 3.4: Better support for smallholder farmers

Affordable access to sustainable phosphorus sources is imperative to ensure food provision for all and to protect the livelihoods of smallholder and marginal farmers. Multiple options exist to help improve phosphorus access in these communities. These include access to credit, extension services, investment in sustainable infrastructure (such as local phosphorus recycling systems from food waste and sanitation where available), and knowledge exchange to support better phosphorus use efficiency and recycling. Developing the capacity to recycle phosphorus from local and regional food systems where available can help to shift reliance away from mineral phosphorus fertilisers.

Smallholder farmers are particularly at risk from volatility in future P prices. Therefore, P stakeholders need to plan for uncertainty and develop adaptation strategies that consider P demand management, including measures targeting increased efficiency in the value chain and selecting for low P footprint nutritious foods. Improving farmer and food stakeholders' adaptive capacity, such as technical know-how, access to equipment and financial resources, provision of extension services and training for smallholder communities and regions to recycle P in local waste streams (see Chapters 6 and 7), may help to diversify P sources and alleviate reliance on expensive mineral P fertiliser. In some cases, vulnerable farmers can adapt to both P and climate exposure through diversification strategies, such as growing more P-efficient and climate-resilient crops or supplementing income with off-farm employment such as agri-tourism (Cordell et al., 2017). Policymakers need to support and enable such measures, to ensure that socio-economic costs of transitioning towards more sustainable practices do not punish the most vulnerable in society.

A substantial barrier obstructing fertiliser use in SSA is farmer poverty, promulgated by the poverty trap in smallholder agriculture (Hanjra et al., 2009). To combat this, government schemes to subsidise fertilisers have been widespread across SSA (AGRF, 2018). At the 2006 Africa Fertiliser Summit, the African Union signed the 'Abuja Declaration on Fertiliser for an African Green Revolution' calling for the elimination of all taxes and tariffs on fertiliser and increased fertiliser use (African Development Bank, 2021). As of 2017, 47 of the African Union states had not achieved this, due to a lack of harmonisation on policy and regulation frameworks, lack of tax incentives, trade barriers, and poor quality control on fertilisers (AGRF, 2018). Ironically, fertiliser subsidies have played a role in this and remain controversial, with evidence of success (Seck et al., 2010), failure (Druilhe and Barreiro-Hurlé, 2012; Jayne and Rashid, 2013), and political misuse (e.g. politically motivated regional allocation of subsidies to win votes) (Banful et al., 2010). However, whilst it is clear that P inputs to nutrient-deficient soils in Africa are urgently needed, this should be delivered in coordination with education and training for farmers to optimise sustainable nutrient practices, especially in terms of appropriate mineral P fertiliser application.

Increasing soil fertility through the addition of nutrients is essential to address soil P deficiency in SSA (Vanlauwe and Giller, 2006). However, mineral fertilisers are not the only means available, and opportunities exist to develop the capacity to fertilise soils using other P sources. Renewable P fertilisers can theoretically be processed from any locally available raw organic material that has a high enough P concentration, contains minimal contaminants, meets local soil and farmer's agronomic needs and can be converted into bioavailable form through cost-effective means (Cordell et al., 2011). Raw sources could include food waste, manure, algae, bones, crop waste and human excreta. Using such sources can also provide co-benefits to environmental and human health in addition to food security. Currently, 54% of the population in SSA does not have access to safe sanitation (WHO/UNICEF JMP, 2017). Aspirational goals to improve sanitation (e.g. the United Nations SDG 6) provide an opportunity to lead global sanitation innovation by building P reuse into sanitation as standard (urine-diverting toilets being one example) (Udert et al., 2016) (see Chapter 7).

Direct application of local phosphate rock with organic materials has also shown to be a successful cheaper alternative to superphosphates on acidic soils (Chianu et al., 2012; Sanchez, 2002). Opportunities that make use of local indigenous knowledge have proven successful, such as Zai pits (small planting pits, usually filled with organic matter to create microenvironments) that reduce soil erosion, retain nutrients (Danjuma and Mohammed, 2015) and optimise plant nutrient uptake from P-rich organic materials. Investment in sustainable infrastructure (such as local P recycling systems from food waste and sanitation) and knowledge exchange to support better P use efficiency and recycling is required. In some cases, extension services and training will be needed to raise awareness of local options for farmers (AGRF, 2018). In other regions, particularly low-income countries and vulnerable communities, increasing affordable access to sustainable P sources will be a priority. Opportunities for achieving this range from access to credit and investment in sustainable infrastructure (such as local P recycling systems from food waste and sanitation where available) to extension services and knowledge exchange to support better P use efficiency and recycling.

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04

Opportunities for better phosphorus use in agriculture

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

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

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

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

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

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

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

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

Challenge 4.4: Recycled phosphorus is not sufficiently used in agriculture

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

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

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

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

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

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

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

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

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

Solution 4.4: Increase phosphorus recycling from manures and residue streams

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

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

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

4.1 Introduction

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

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

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

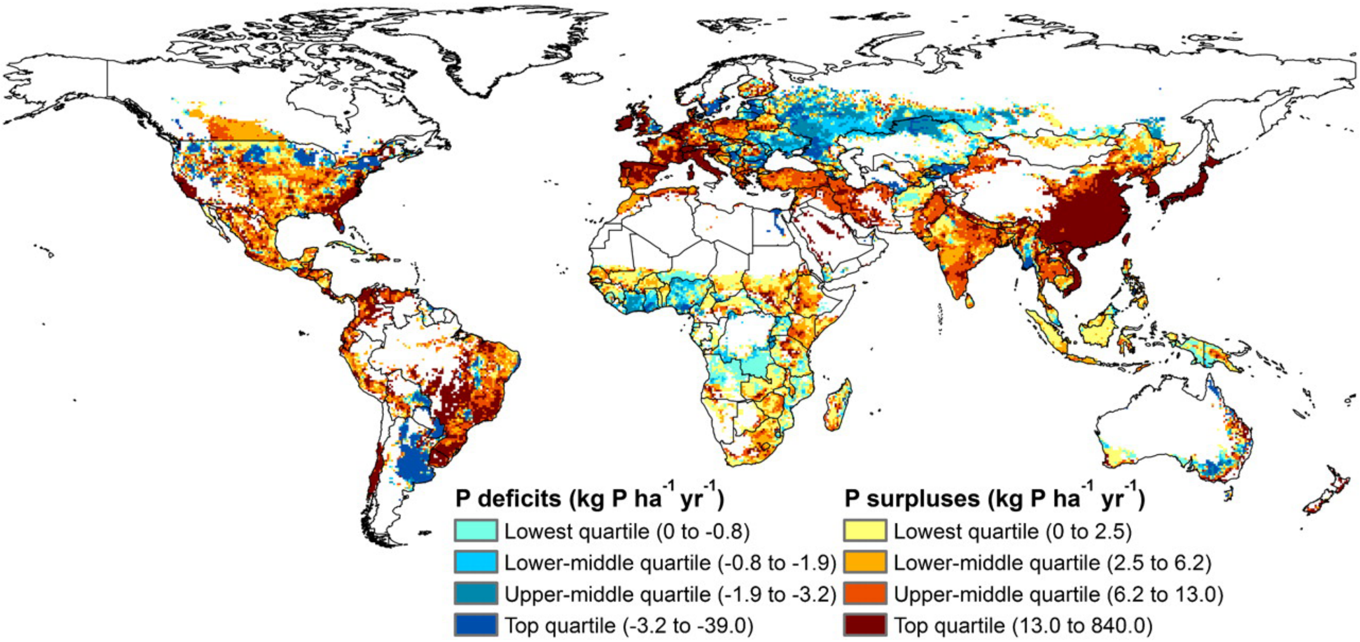

Figure 4.1 Global map of agronomic phosphorus (P) imbalances for the year 2000 expressed per unit of cropland area in each 0.5° grid cell. The P surpluses and deficits are each classified according to quartiles globally (0–25th, 25–50th, 50–75th, and 75–100th percentiles). Figure copyright of MacDonald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

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

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

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

4.2 Phosphorus flows in the global agricultural system

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

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

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

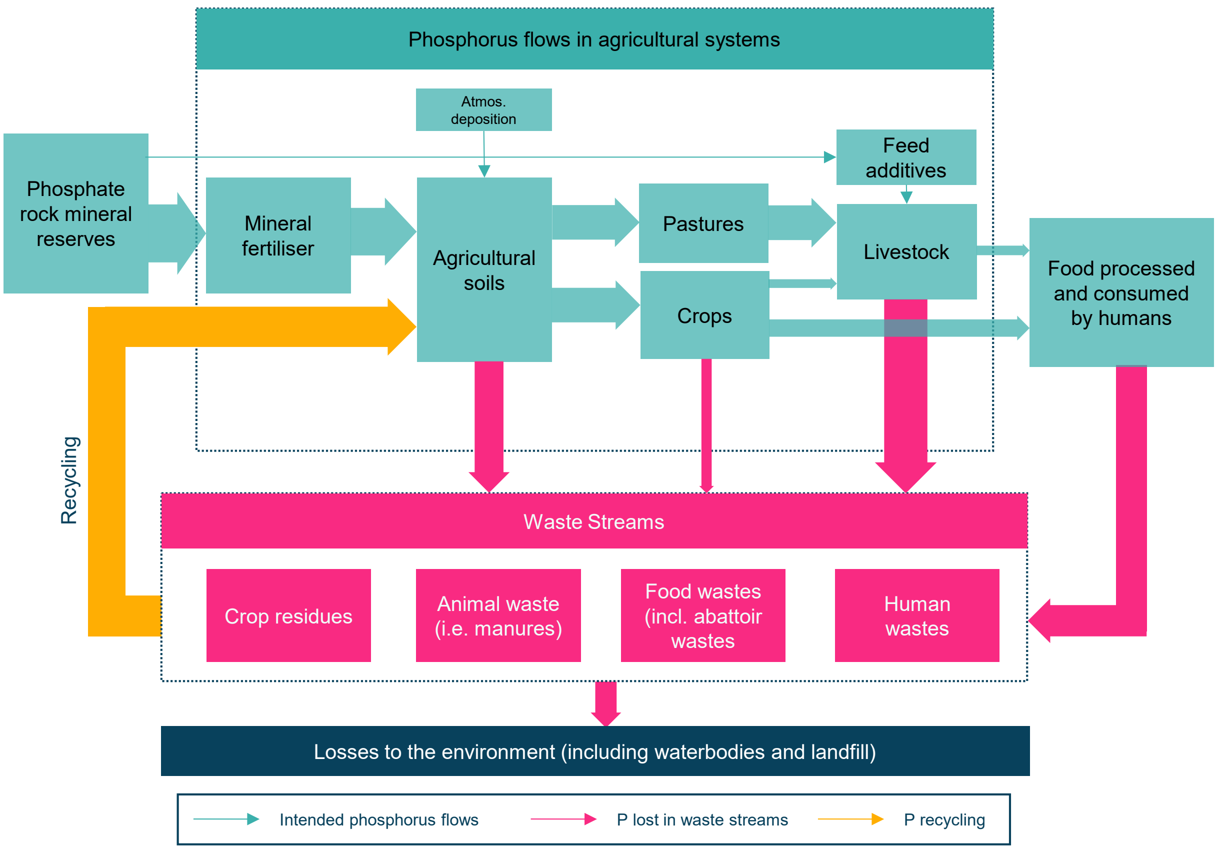

Figure 4.2 A conceptualisation of the phosphorus (P) flows to, within and out of agricultural systems (not including aquaculture and forestry). The width of arrows is proportional to the amount of P estimated in each flow and is based on data from Chen and Graedel (2016). However, wide variation not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

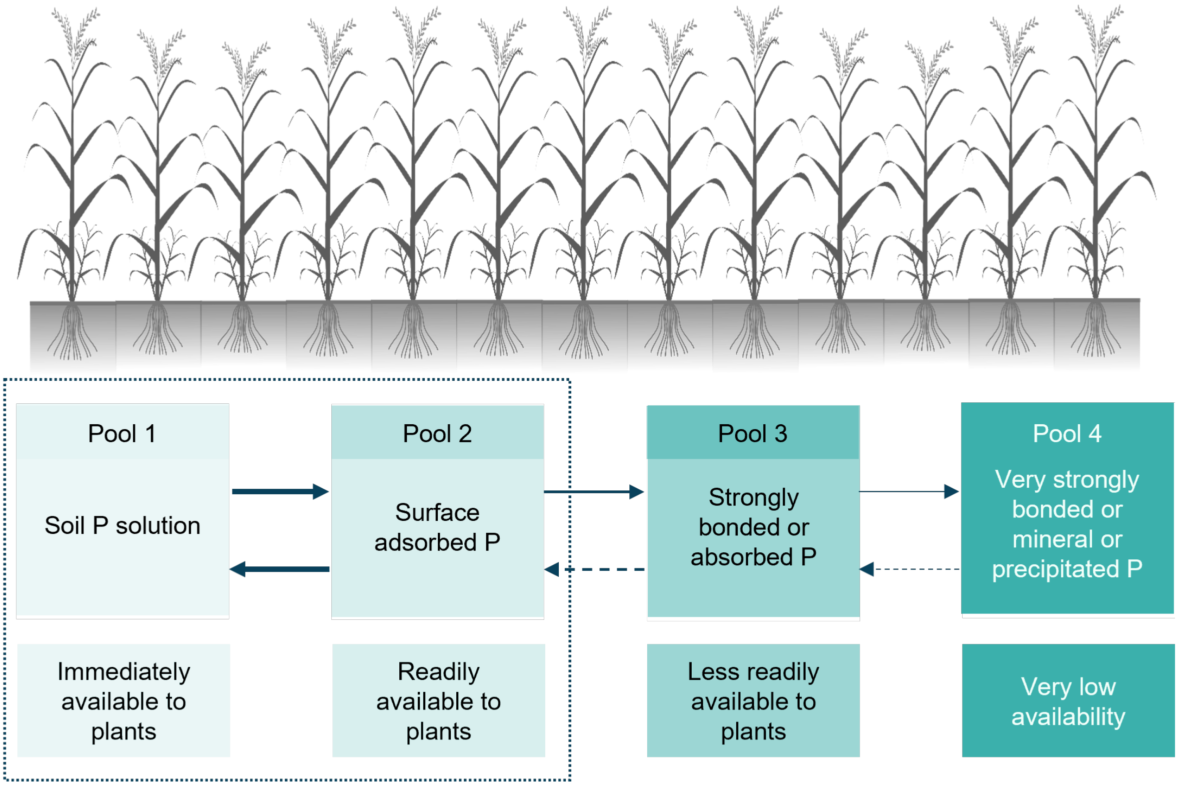

Figure 4.3 Conceptual diagram showing interactions between the forms of inorganic phosphorus (P) in soils categorised in terms of accessibility and plant availability (modified from Syers et al. (2008)).

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

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

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

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

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

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

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

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

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

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

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

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

4.3 Phosphorus budgets and use efficiency in agriculture

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

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

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

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

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

Authors: Heidi Peterson and Tom Bruulsema

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

Defined as the equation,

PUE = crop removed/P source inputs

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

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

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

4.4 The win-wins of better phosphorus management in agriculture

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

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

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

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

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

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

4.5 Challenges

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

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

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

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

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

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

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

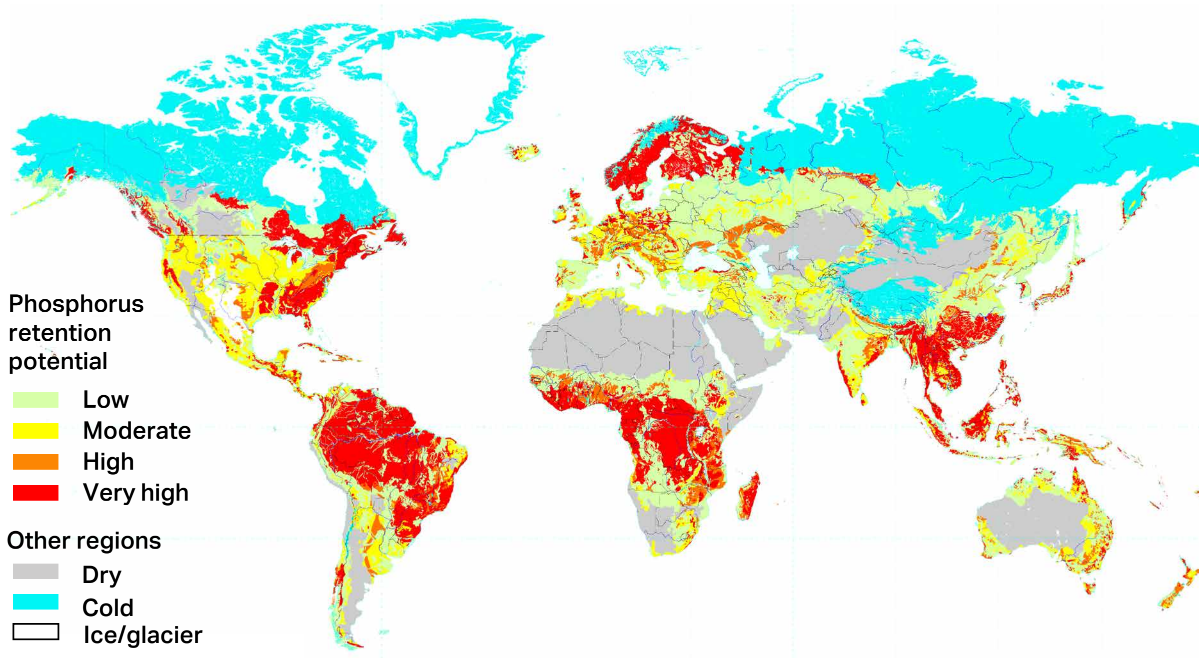

Figure 4.6 Spatial variation in the soil phosphorus retention (adsorption) capacity across the world, indicated by colour, from low (green) to very high (red) (image courtesy of the USDA-NRCS (1998)).

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

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

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

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

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

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

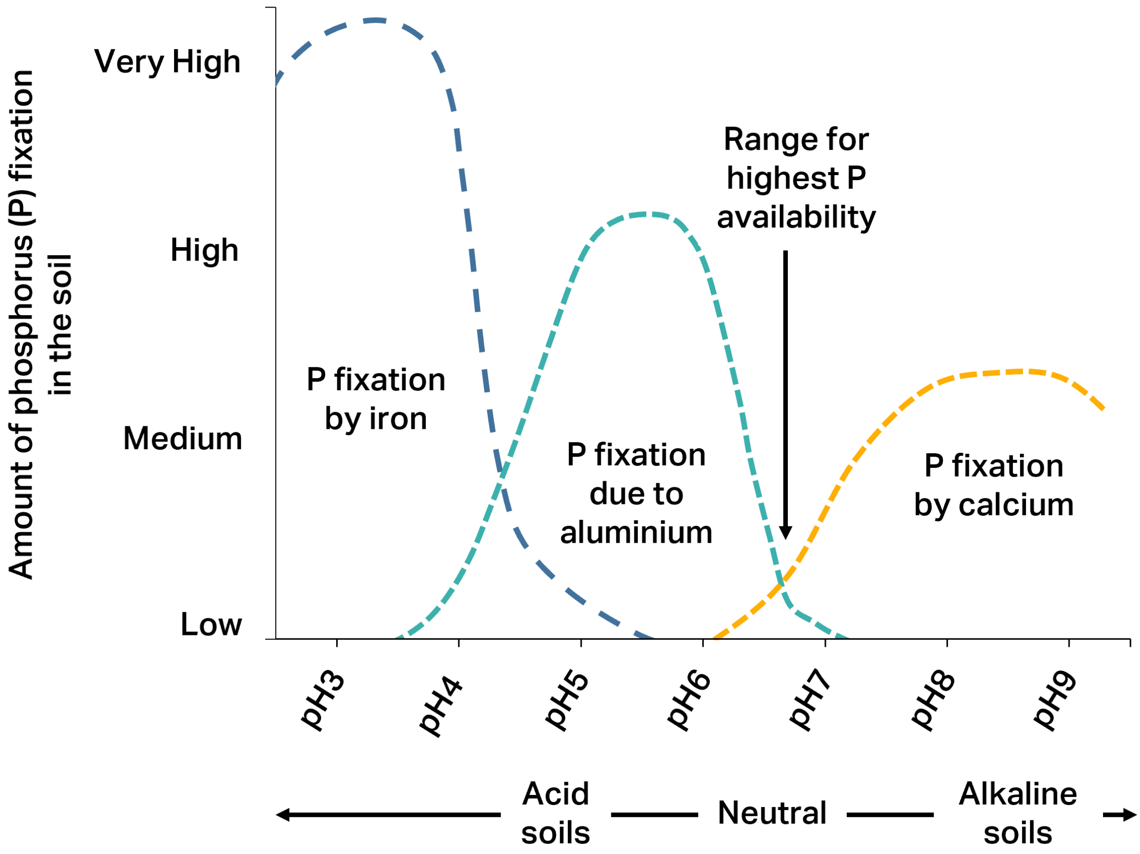

Figure 4.7 General qualitative representation of soil phosphorus (P) fixation (and hence availability) as impacted by soil pH. Modified from Price (2006).

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

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

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

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

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

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

Challenge 4.4: Recycled phosphorus is not sufficiently used in agriculture

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

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

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

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

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

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

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

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

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

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

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

4.6 Solutions

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Solution 4.4: Increase phosphorus recycling from manures and residue streams

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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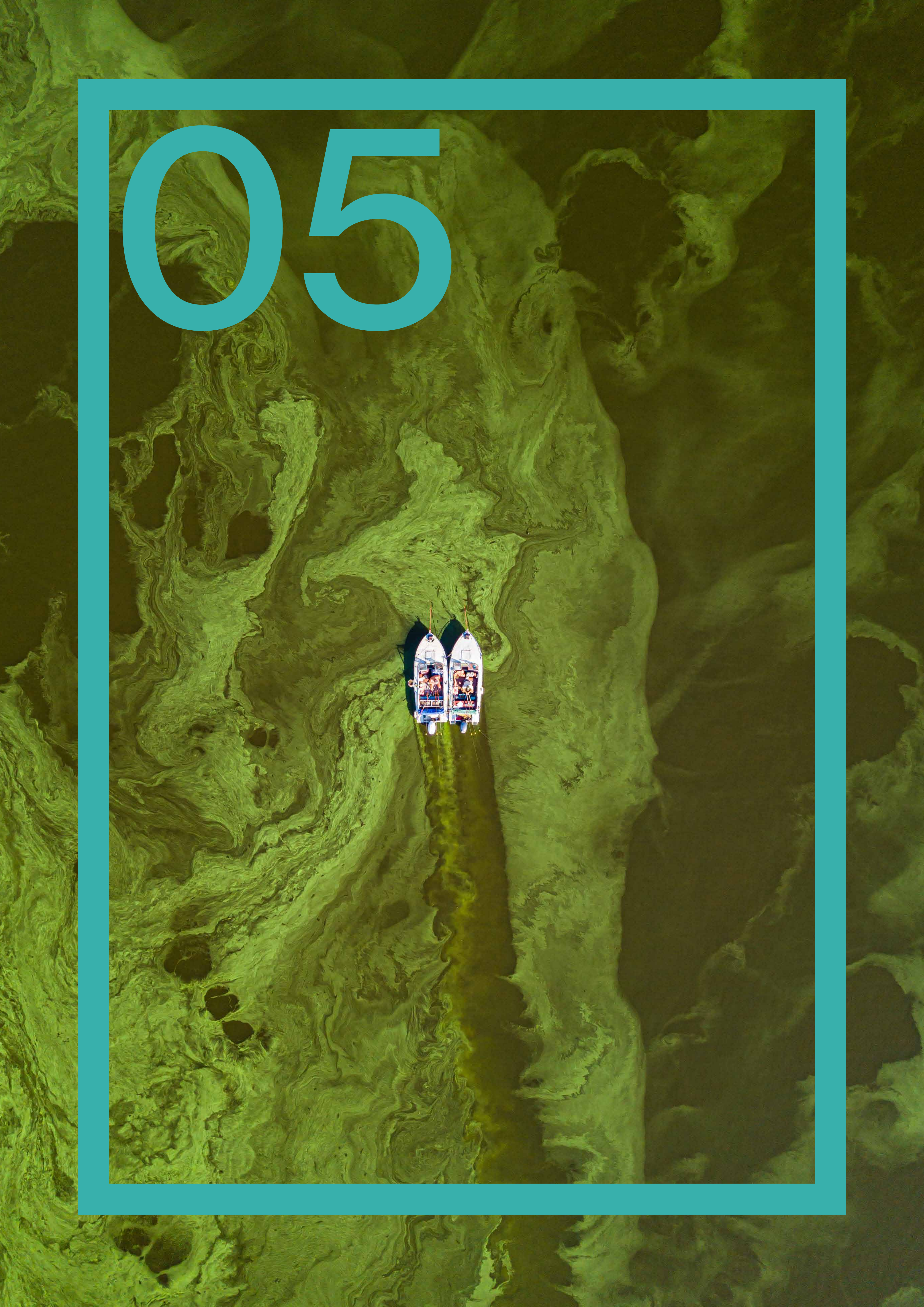

Phosphorus and water quality

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Left: Two boats floating in an algal bloom in Labelle, Florida, triggered by elevated phosphorus concentrations. Photograph courtesy of Adobe Stock.

Phosphorus is one of the key drivers of the global nutrient challenge and the biodiversity loss emergency with respect to freshwater and marine ecosystems. Impacts include toxic algal blooms, mass fish kills, greenhouse gas emissions, and the loss of economic, societal, and cultural value associated with high-quality ecosystems. The 'know-how' to deliver significant water quality improvements across sectors and scales is available, and many of the solutions provide multiple benefits. The challenge now lies in mobilising policymakers, investment, and public support for change.

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Challenge 5.1: Phosphorus pollution is increasing globally

Over the course of the 20th-century, phosphorus losses from land to fresh waters almost doubled because of human activity. Whilst sources of phosphorus pollution vary between regions, they are dominated by agricultural (e.g. livestock manures and fertilisers) and wastewater discharges. In many regions, phosphorus losses continue to increase.

Challenge 5.2: The global impacts of phosphorus pollution are not well quantified

Elevated phosphorus concentrations in freshwater and coastal marine ecosystems are contributing to the unprecedented loss of freshwater biodiversity and the growing global phenomenon of freshwater and marine 'dead zones'. However, the true scale of the problem is difficult to estimate as baseline data are lacking across all regions and scales. Long-term monitoring programmes are necessary to track and study recovery following nutrient reduction strategies and to inform adaptive management initiatives.

Challenge 5.3: Phosphorus losses and their impacts are expensive

The direct and indirect impacts of eutrophication are costly, in terms of losses of ecosystem services, clean up expenses, and losses to local economies. Phosphorus losses also represent a significant waste of resources. Global or regional assessments on the costs of eutrophication or the effectiveness of measures to reduce phosphorus losses are lacking. This severely compromises the ability to communicate the need for action with stakeholders and policymakers.

Challenge 5.4: There is a lack of phosphorus policy and legislation covering water security

Phosphorus sustainability is not consistently enacted in regional policies and global action is needed to bring phosphorus enrichment of waters to the attention of policymakers. No global holistic policy on nutrient management in aquatic ecosystems exists. A key challenge is therefore enabling better integration of a sustainable phosphorus strategy across existing and emerging policy frameworks.

Solution 5.1: Reduce phosphorus losses and improve phosphorus use efficiency

Improved agricultural and wastewater management should be implemented to reduce losses of phosphorus from land to water. There is also a clear opportunity to improve phosphorus use efficiency in aquaculture. In order to reduce phosphorus pollution on a global scale, we must identify opportunities to decrease the amount of 'mined' phosphorus entering the anthropogenic phosphorus cycle, enhance uptake of sustainable fertiliser management approaches, and take action to close the phosphorus loop. This can be done by cutting phosphorus losses and increasing recycling and phosphorus storage within the landscape.

Solution 5.2: Implement new and utilise existing data collection systems to inform adaptive management

Monitoring programmes provide a critical link between information, evidence-based decision making, and policy development, and should be used to inform adaptive management frameworks. This is especially important given ecosystem restoration is often a long-term process, and considering the impacts on waterbodies of multiple stressors, including those associated with climate change, population growth, and urbanisation. Restoration efforts must be coupled with preventative interventions to safeguard those ecosystems that are sensitive to future increases in phosphorus input.

Solution 5.3: Implement integrated catchment management and develop algal bloom response plans

Integrated phosphorus management strategies that cross scales will be essential in achieving improved water security globally. A road map for capacity development is required to support the wider development of long-term integrated catchment management programmes focused on phosphorus. Rapid response plans are needed to manage the risk of damage to both ecosystem and human health associated with harmful algal blooms.

Solution 5.4: Develop integrated policy approaches and globally coordinated phosphorus initiatives

Solutions to overcoming phosphorus inefficiencies must rely on tackling phosphorus imbalance at all scales. The development of regional targets, mandates and incentives are essential, and will often require transboundary cooperation. Where regional policies exist on phosphorus or other nutrients, experiences with these should be synthesised to inform their improvement as well as support policy development in other regions where no relevant policies exist.

5.1 Introduction

The enrichment of fresh and coastal waters with nutrients including phosphorus (P) and nitrogen (N) is one of the most conspicuous impacts of the Anthropocene (Smith and Schindler, 2009). That we continue to pollute the very water that we rely on for survival is a shocking level of self-harm. That we are willing, by our actions, to cause alarming rates of biodiversity loss in fresh and coastal waters is equally shocking. The rate of biodiversity loss in fresh waters is higher than in any other planetary domain (Tickner et al., 2020). Over 25% of all freshwater species are currently threatened with extinction globally, and freshwater fauna declined globally by 83% from 1970 to 2014, compared to 60% for all habitat types (WWF, 2018; Reid et al., 2019; Tickner et al., 2020). While a wide range of emerging and persistent stressors are driving these losses, climate change and increasing nutrient delivery from food production and consumption are ubiquitous. They combine to generate a globally increasing incidence of eutrophication, the process whereby excess input of nutrients (N and P) drives the formation of harmful algal blooms, coastal dead zones, mass mortalities of fish, closure of economically important fisheries and shell-fisheries, high rates of biodiversity loss, high rates of greenhouse gas emissions, and the loss of economic, societal and cultural value associated with high-quality ecosystems. The process of eutrophication in lakes through P loading has become a central exemplar of the links between ecological behaviour and natural capital and economics (Dasgupta, 2021). Beyond lakes and fresh water, in only a few thousand years, the contribution of P towards long term ocean anoxia has potentially unthinkably damaging consequences to the Earth's biogeochemistry (Watson et al., 2017).

In the this chapter, we highlight the global nutrient challenge and biodiversity loss emergency with respect to freshwater and marine ecosystems and define the role of P as one of the key drivers of this problem. We highlight the importance of balancing P losses alongside N losses, predominantly from food systems and human waste, to relieve the effects of eutrophication. We stress that the notoriously difficult task of restoring ecosystems is within reach. However, the economic and cultural costs of large-scale environmental management are likely high and should be equitably shared. It is, therefore, important to raise awareness of ecosystems under threat and to work across governments to ensure their long-term integrity, as well as to identify short-term disaster response plans where trends of degradation are deemed unacceptable. We call for a greater focus on preventative nutrient management to safeguard global biodiversity in freshwater and coastal ecosystems and meet long-term sustainability goals.

To overcome P losses, we must tackle P imbalance across scales and not just those on the farm (Shepherd et al., 2016). Critically, this requires that eutrophication control strategies reduce whole system total P inputs (Figure 5.1), particularly to the food system, whilst maintaining or increasing production outputs and increasing P use efficiency within the system (Withers et al., 2014). Transdisciplinary approaches to sustainable P management that embrace both field-scale and wider regional P stewardship, that allow for variance in the response of damaged ecosystems to management and

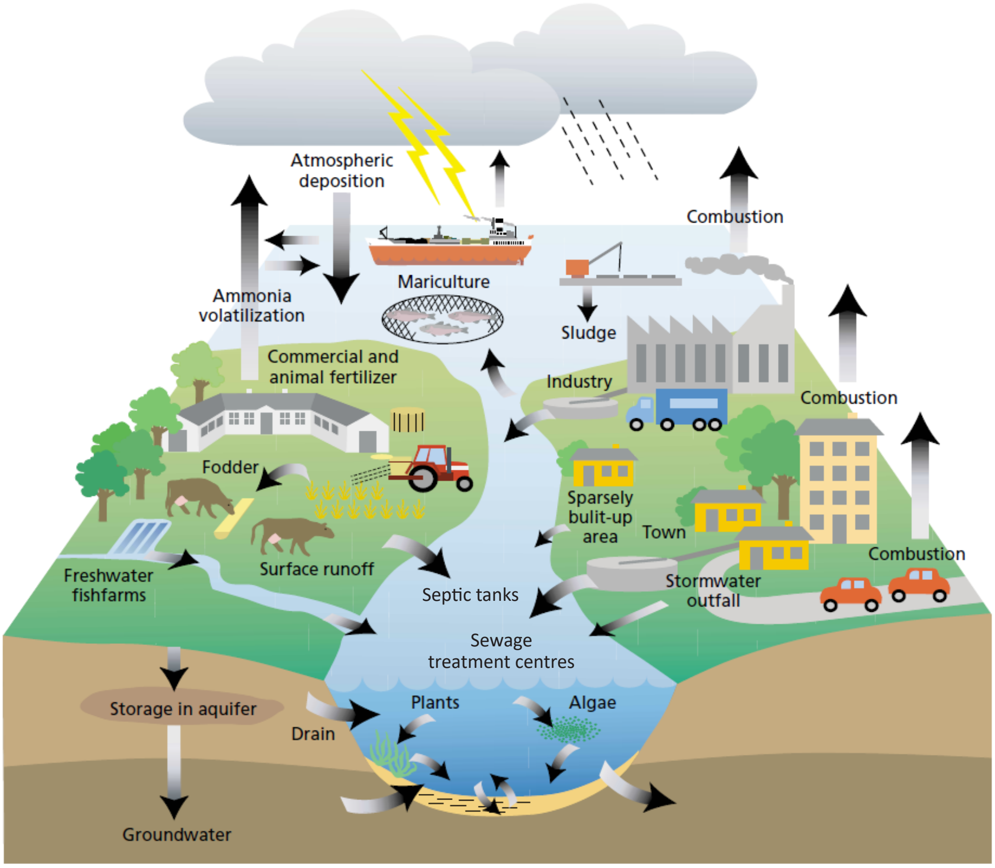

Figure 5.1 Anthropogenic sources of phosphorus to the aquatic environment. The main anthropogenic nutrient sources are discharges from agricultural runoff, sewage treatment works and industry. Modified from Ærtebjerg et al. (2003).

control the socio-economic drivers for change have been proposed (Carpenter et al., 2015; Jacobs et al., 2017; Withers et al., 2018). These approaches include the need to address nutrient imbalances in agricultural development, globally (Vitousek et al., 2009). Opportunities to develop a circular economy to reduce new imports of inorganic P into existing farming and food processing systems (e.g. Metson et al., 2016) (see Chapter 8), and recovery of P from different wastewaters to reduce direct effluent loadings to rivers and lakes (see Chapter 7) are needed. The potential to reduce society's P demand by altering dietary choice (see Chapter 8), reducing food waste and genetic design of crops (see Chapter 4) should be considered (Johnes, 2007a; Withers et al., 2018). Reducing total societal demand for P would have a positive cascading effect, reducing P inputs and losses across the food-value chain (see Chapter 4). This lowering of P surpluses would lead to a rebalancing of nutrient inputs and outputs of P in agricultural systems, with lower landscape P accumulation, which would both reduce P losses to water in the longer term and increase catchment P buffering capacity. We develop the evidence base on ecosystem responses to these measures and highlight the need to consider them holistically, across large catchments, and across political divides in order to deliver large-scale environmental and socio-economic gains.

5.2 Impacts of phosphorus on freshwater and coastal ecosystems

Phosphorus concentrations in aquatic ecosystems have been elevated worldwide by human activities. For example, in the USA, P concentrations in 72% of rivers and 79% of lakes exceeded background levels because of human activity in the last decades (Dodds, 2006). In the European Union (EU), ~32,000 km^2 lake surface area (about 40% of monitored lakes by number) is deemed to fail ecological quality targets under the EU Water Framework Directive (European Parliament, 2000). Over 83% of freshwater habitats in the EU were classed as being in an unfavourable condition in 2015, higher than any other habitat type (European Environment Agency, 2015), many of which are impacted by eutrophication. Over 50% of the P mass input (load) to 23 of the world's largest lakes originates from human activities representing a threat to current and future water security (UNEP, 2016). Water security has been defined as "The capacity of a population to safeguard sustainable access to adequate quantities of acceptable quality water for sustaining livelihoods, human well-being, and socio-economic development, for ensuring protection against water-borne pollution and water-related disasters, and for preserving ecosystems in a climate of peace and political stability" (UN-Water, 2013). Strong evidence indicates that eutrophication effects are increasing in poorly monitored areas of the world. For example, in China, eutrophication in the previous century was reported to have contributed to the elimination of all fish from about 5% of river length (Dudgeon, 1999). In the coming decades, increasing P loadings to aquatic ecosystems threaten the Chilean Lake District, a globally important biodiversity hot spot (Pizarro et al., 2010; Almanza et al., 2019), adding to the catastrophic algal blooms already impacting the southern Chilean coast, a significant region for finfish mariculture (Bouwman et al., 2013a).

Due to its relative scarcity in bioavailable forms, P is often the key nutrient limiting or co-limiting plant growth in fresh waters (Carpenter et al., 2005; Smith and Schindler, 2009; Smeti et al., 2019; Mackay et al., 2020). This has led to wide-reaching directives and policies with a strong focus on reducing P pollution in countries and regions where eutrophication effects have developed over the last century (e.g. the USA Clean Water Act, China's Law on Water Pollution Prevention and Control and the EU Water Framework and Habitats Directives). In some cases, such as the USA Clean Water Act, efforts have been focused on P point sources, neglecting diffuse sources. However, even where significant P reductions have been achieved these have not necessarily led to expected ecological improvements (e.g. Sharpley et al., 2013; Carvalho et al., 2019).

It is now known that degraded ecosystems can resist recovery, meaning that ecological responses may be reliant on the reduction of nutrients to below pre-impact conditions (Scheffer and Nes, 2007; Smith and Schindler, 2009; Ibáñez and Peñuelas, 2019) and that other stressors, including other nutrient stressors, climate change and invasive species, may modify the effects of P reduction and inhibit or delay ecological recovery (Moss et al., 2011). Phosphorus reduction, in many cases, may not have gone far enough to support ecological improvements (e.g. in the Baltic Sea; Ollikainen et al., 2019). Phosphorus stored within catchments, aquifers, and bed sediments during decades of enrichment, termed 'legacy P', can be released back to the water, delaying recovery for many years following a reduction in catchment P loading (Sharpley et al., 2013; Haygarth et al., 2014; Steinman and Spears, 2020). Whilst there have been some promising examples of ecological recovery following P (and N) loading reductions (e.g. Jeppesen et al., 2005b; Bowes et al., 2011; Riemann et al., 2016; Schindler et al., 2016) we stress caution on reports of 'global scale reversal of eutrophication' (Ibáñez and Peñuelas, 2019) based on large-scale nutrient load reductions where evidence of ecological recovery has not also been clearly demonstrated.

Many freshwater and coastal ecosystems in human-altered landscapes are currently experiencing low levels of stress, making them highly sensitive to any future increases in nutrients. In the EU, for example, about 70% of lake surface area, assessed under the EU Water Framework Directive, is judged to have low-moderate levels of stress that are currently judged to be acceptable (Spears et al., 2021). These ecosystems and others are highly sensitive to any further environmental change including the effects of climate change. Some lakes are experiencing warming of up to 0.7 °C per decade (O'Reilly et al., 2015), in addition to increased frequencies and magnitude of floods and droughts (IPCC, 2019), which will affect the biological response to, and impact on, nutrient cycling within lake ecosystems (Moss, 2010; Steinman and Spears, 2020). Such extreme events have been shown to cause rapid losses of biodiversity and ecosystem integrity, for example, following hurricanes in Lake Apopka, Florida, USA (Havens et al., 2001), and may also act to flush out accumulated nutrient stores in wetlands and rivers, resetting the baseline nutrient status of these systems (Johnes et al., 2020) and enriching downstream reaches, estuaries and coastal waters. Without the development of novel preventative management approaches, which may include stricter nutrient reduction targets to mitigate climate change effects on water quality (Spears et al., 2021), the burden of restoration will increase on future generations (Damania et al., 2019).

Eutrophication is commonly associated with a shift from rooted aquatic vegetation towards bloom-forming algae in the water column (Sayer et al., 2010) and is responsible for the global proliferation of toxin-producing cyanobacteria, also called harmful algal blooms (HABs) (Paerl and Paul, 2012) (Figure 5.2). An increase in algal biomass can lead to increased biological oxygen demand causing oxygen depletion which can restrict habitat for fish (Hendry et al., 2003; Foley et al., 2012). In extreme cases, these conditions can lead to catastrophic mass mortalities (Chen et al., 2009) (Figure 5.2).


a)
b)
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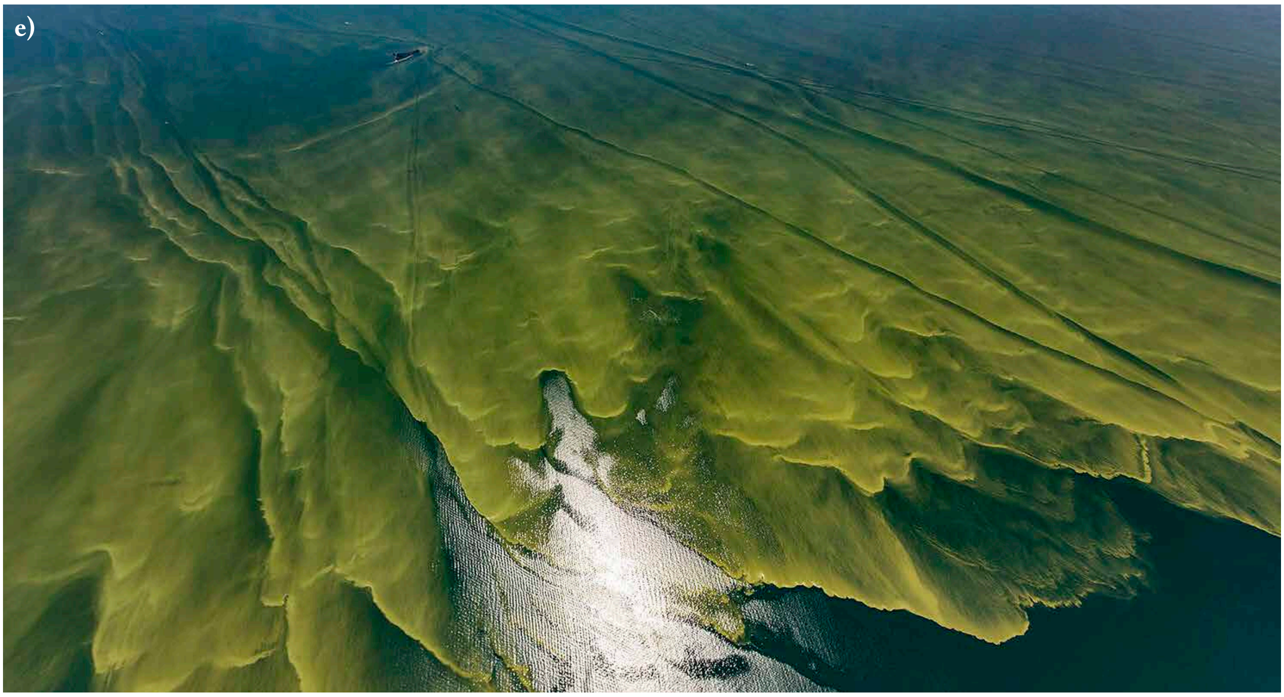

f)

Figure 5.2 Deleterious effects of eutrophication.
a) An algal bloom in Dianchi Lake, China in 2007. Despite millions spent to clean up the lake, the water remains undrinkable and unfit for agricultural or industrial uses. Photo Credit: Greenpeace China.
b) Satellite image of an algal bloom in the Baltic Sea (approximately 290 km wide by 390 km long). Photo credit: European Space Agency.
c) Soldiers clear algae along the coastline of Qingdao, Shandong province, in 2008. More than 10,000 people and 1,200 vessels were mobilised to tackle the huge algae bloom that threatened the Olympic sailing event in Qingdao. Photo Credit: Asianewsphoto - Ju Chuanjiang.
d) An aquaculture farmer cleans away dead fish at a lake in Wuhan, China, 2007. More than 110,000 pounds of fish died due to phosphorus pollution and hot weather in the lake. Photograph courtesy of China Daily/Reuters.
e) Huge harmful algal blooms float towards the coastline of Lake Erie, US, in 2017. Photo Credit: Aerial Associates Photography, Inc. by Zachary Haslick.
f) A fisherman sets out nets to catch fish in a river heavily polluted by phosphorus in West Bengal, India. Photograph taken by Apratim Pal - https://www.instagram.com/guycalledapratim/?utm_medium=copy_link.

Increased water turbidity during algal blooms reduces light penetration and inhibits photosynthesis by rooted aquatic plants (Hautier et al., 2009). This negatively impacts highly sensitive and diverse littoral, or shallow water, habitats (Scheffer and Nes, 2007; Penning et al., 2008) and contributes to the highest rates of biodiversity decline across all ecosystems. The Living Planet Index tracks the state of global biodiversity by measuring the population abundance of thousands of vertebrate species around the world; the Freshwater Index has declined by 83% between 1970 and 2014 (WWF, 2018). In extreme cases, which are unfortunately common, aquatic plants can die-off completely (Sayer et al., 2010), removing valuable habitats and food for invertebrates, fish, and wild bird species (Rönkä et al., 2005).

Under very high P concentrations, lakes can be turned into near monocultures of harmful cyanobacteria (O'Neil et al., 2012; Paerl and Paul, 2012). Some cyanobacteria, such as *Microcystis aeruginosa* and *Dolichospermum spiroides*, release neurotoxins and hepatotoxins. The collective term for the family of toxins produced by cyanobacteria is 'cyanotoxins'. Cyanotoxins are harmful to mammals, causing deaths of livestock (Briand et al., 2003) and dogs (Backer et al., 2013), and represent a risk to human health through consumption of contaminated water and food (Codd et al., 2005), and potentially through the dispersal of aerosols (Facciponte et al., 2018).

Although epidemiological data remain sparse, the human health risk associated with cyanotoxins (Figure 5.3), and the role of P and N in increasing this risk, has long been recognised by the World Health Organization (WHO, 1999).

Evidence from an analysis of European lakes suggests that reduction of P concentration may be key in reducing this risk (Figure 5.4; Carvalho et al., 2013). Humans may be exposed to cyanotoxins through ingestion of untreated drinking water and direct contact with water during recreation (see Kubickova et al. (2019) for a review of human health reports globally). Chronic and long-term exposure through food represents a largely unquantified exposure route, especially concerning the consumption of freshwater and marine fish and shellfish produced within eutrophic ecosystems (Huang and Zimba, 2019). In addition, the nutrient status of waters may increase the abundance, composition, virulence, and survival of pathogens that are already present in waterbodies, increasing the risk of the spread of infectious diseases where waterbodies undergo nutrient enrichment (Smith and Schindler, 2009).

The effects of food preparation methods on cyanotoxin content are not well understood. Growing evidence suggests a gene/environment interaction through consumption of food containing the cyanotoxin beta-methylamino-L-alanine (BMMA) linked to early-onset of neurodegenerative diseases including Alzheimers and Parkinson's disease in Guam, with human environmental exposure to BMAA proposed to be widespread (Holtcamp, 2012), but currently unquantified. The microbial decomposition of cyanobacteria in surface waters and the stimulation of geosmin and 2-methylisoborneol (MIB) production by algae and bacteria in response to nutrient enrichment can result in taste and odour problems in drinking water supplies and can taint fish and shellfish caught for human consumption, generating significant clean-up costs for the water industry (e.g. Parinet et al., 2010; Davidson et al., 2014a).

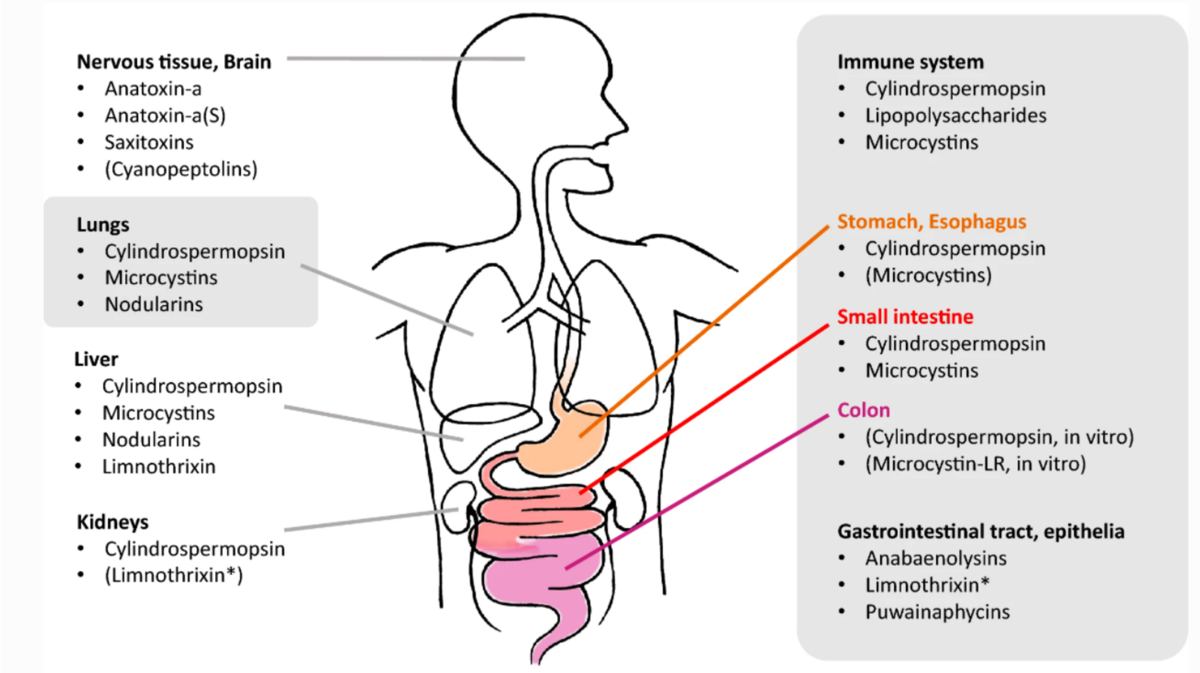

Figure 5.3 Organs/organ systems in humans affected by toxic metabolites of cyanobacteria. Organs traditionally considered targets of toxicity are on the left; organs directly subjected to oral exposure are on the right. Grey-shaded boxes highlight organs with mucosal surfaces that serve as primary entry portals for environmental and dietary contaminants. Evidence is based on a review of 29 exposure case studies with a global spread. Image courtesy of Kubickova et al. (2019).

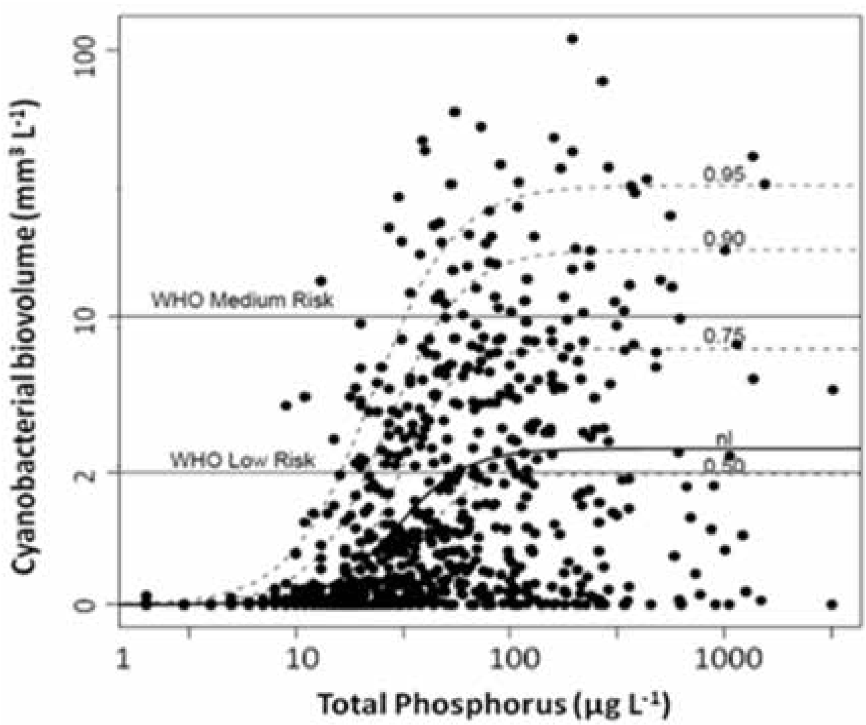

Figure 5.4 Relationship between cyanobacteria biovolume and total phosphorus concentration in > 1500 European lakes in relation to the World Health Organization (WHO) Risk Thresholds for managing human health risk in recreational waters (WHO, 1999). At cyanobacteria biovolumes of between the Low to Medium Risk thresholds, those in contact with water should expect a greater risk of short-term health effects including skin irritations and gastrointestinal illness. Humans in contact with waters with cyanobacteria biovolumes above the Medium threshold but below the High threshold (contact with surface scums) are more likely to experience longer-term illnesses and a greater risk of skin irritations and gastrointestinal illness. Above the High-Risk threshold (not shown in this figure), exposure may also result in acute poisoning, especially where underlying health issues are present, such as those being treated for renal disease. Image source: Carvalho et al. (2013).

Algal blooms can significantly increase filtration costs in potable water treatment works, and in some cases make water unsuitable for drinking (Qin et al., 2010; Agrawal and Gopal, 2013). Another issue for food is the application of cyanotoxin-laden irrigation water to crops, which may result in bioaccumulation of microcystin congeners with associated human health risks (Cao et al., 2018a; b). Phosphorus management and control of access to waterbodies can be used to reduce the likelihood of exposure in fresh waters (Huisman et al., 2018), but brings consequences of reducing amenity and recreational use of waters. In coastal ecosystems, where deaths of large aquatic animals, including turtles and manatees (Capper et al., 2013), dolphins (Fire et al., 2015), and whales (Vos et al., 2003), have been linked to algal toxin ingestion, the direct role of P is less clear.

We know that P contributes, with N, to the creation of marine 'dead zones' (Diaz and Rosenberg, 2008; Rabalais et al., 2010), but the full extent of ecological responses in coastal ecosystems to P pollution, and the likelihood of addressing these problems through nutrient reductions alone, has been questioned (Duarte and Krause-Jensen, 2018). Although nutrient delivery to some coastal ecosystems may be driven by marine sources, through upwelling or advection (Anderson et al., 2002) (Anderson et al., 2008), the main cause of the increase in coastal eutrophication at a global scale has been attributed to nutrient losses from land-based sources, including fertilisers applications (for both N and P), and fossil fuel combustion (for N only) (Howarth, 2008). Coastal ecosystems appear particularly vulnerable to eutrophication, with impacts reported on fisheries, shell-fisheries, amenity, recreation, and coastal defences (Rabalais et al., 2009; van Beusekom, 2018), and global-scale loss of seagrass habitats that mirror the loss of vegetation in lakes (Deegan et al., 2012).

The scale of this challenge is becoming clearer. Over 400 coastal ecosystems have been reported globally as 'dead zones', 13 of which are classified as recovering, affecting a total area of more than 245,000 km^2, and resulting in the loss of high-value biodiversity and fish stocks; coastal ecosystems in China, Europe and the USA appear most at risk from eutrophication (Diaz and Rosenberg, 2008; Rabalais et al., 2010). Whilst N is usually considered the dominant limiting nutrient in coastal waters, P is also important in highly enriched or enclosed ecosystems and multiple stressor effects are common (Howarth and Paerl, 2008; Conley et al., 2009).

Nutrient management approaches for both N and P on land should be developed to deliver benefits across freshwater and coastal ecosystems (Paerl et al., 2016), an approach that is acknowledged by the UNEP International Resource Panel with respect to reducing impacts of land-based activities on the Blue Economy (IRP, 2021)

5.3 Phosphorus sources and transport from land to sea

The reduction of P concentrations in aquatic ecosystems as part of an integrated nutrient management strategy lies at the core of control of freshwater eutrophication globally. To achieve this, it is often first necessary to identify anthropogenic sources of P within the catchment and to understand how they are transported and transformed within the water system, alongside N sources and fluxes (Figure 5.1; Anderson et al., 2002; Neal and Heathwaite, 2005; Hilton et al., 2006; Sharpley et al., 2013). Sources are often distinguished as 'point' or 'diffuse' in origin. Nutrients from point sources (e.g. domestic and industrial wastewater discharges and sewage overflows and organic wastes discharged from pit latrines) enter the water at a specific site and are often persistent and continuous in delivery. This can make point sources easier to locate, monitor, and manage when compared to diffuse sources.

Diffuse sources of P originate from activities that do not have one discrete source and are often episodic in delivery (Heathwaite and Johnes, 1996; Edwards and Withers, 2008), frequently as a result of heavy rainfall (Johnes, 2007b; Stutter et al., 2007; Outram et al., 2014). Common diffuse sources of P include runoff from agricultural land with high applications of fertiliser, manures and slurries, pastures grazed by livestock, runoff from roads and construction sites (particularly in agricultural landscapes) and recreational areas treated with fertiliser (e.g. golf courses, lawns and gardens). These typically mobilise P in a soluble form and are attached to eroding agricultural soils as particulate P.

Typically, diffuse sources of P are delivered to waterbodies through overland flow in storm events and after periods of prolonged rainfall, through the soils as throughflow, and more slowly via groundwater flow pathways (i.e. leaching) following the over-addition of P-based fertilisers (Bingham et al., 2020). Phosphorus delivery events are often associated with runoff from soils with high P content, which may have been generated anthropogenically by the historic and current application of fertilisers and manure (Ærtebjerg et al., 2003; Haygarth et al., 2012). Collectively, the mobilisation and transport of P to waters are conceptualised as a transfer continuum (Figure 5.5; Haygarth et al., 2005). This describes P transport from land to sea from sources in the landscape (e.g. fertiliser, animal feed, natural soil levels), mobilisation (e.g. from soil, as solubilisation in chemical leachate, detached with particles or with accompanying incidental losses of freshly applied surface deposits), delivery (e.g. hydrological transport through or over the landscape) and finally impact (e.g. economic or ecological, which can occur over 100s of km and many years after the start of the continuum).

Nutrient retention capacity is considered the sum of a landscape's capacity to remove P from solution and through trapping of particulate P in transit from land to stream (Heathwaite et al., 2005; Schippers et al., 2006; Heckrath et al., 2007; Shigaki et al., 2007; Johnes et al., 2007). Due to its ability to bind P, soil type exerts a significant control on P retention and mobilisation in a landscape (Shen et al., 2011). Importantly, metal cations of

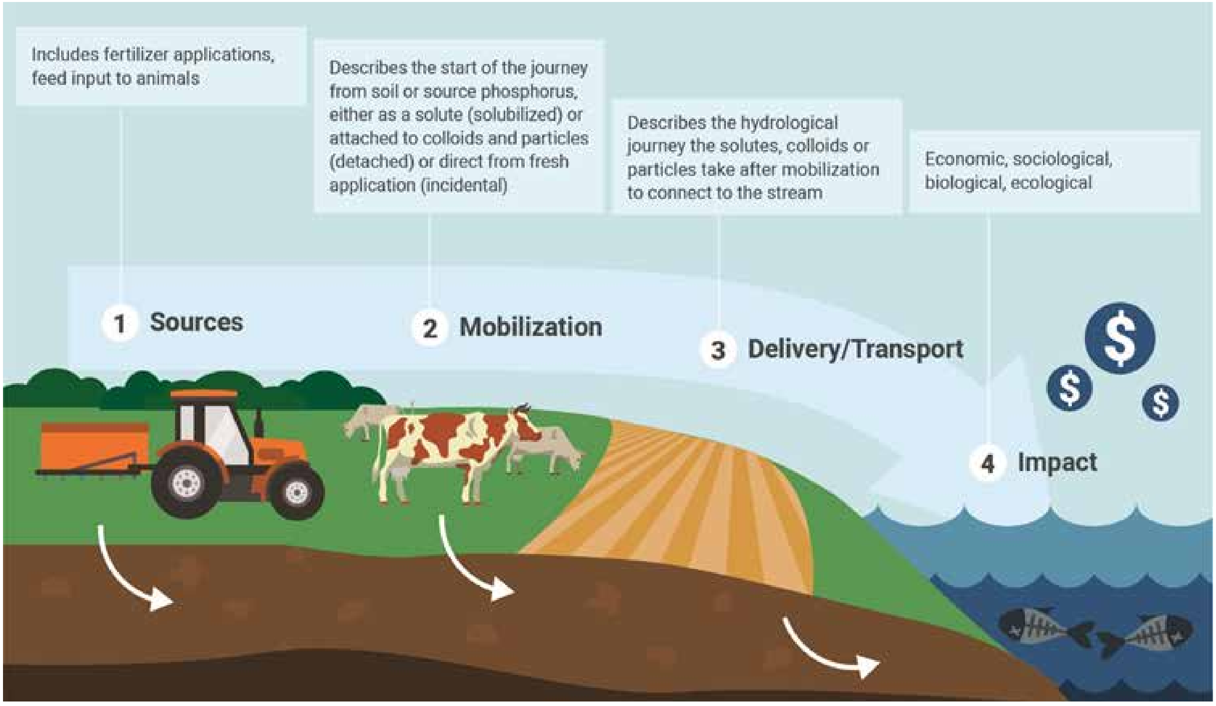

Figure 5.5 The phosphorus (P) transfer continuum. Soil erosion by water has been estimated to contribute to over 50% of total P losses (Alewell et al., 2020), though this can vary markedly between catchments both spatially and temporally (Johnes, 2007b; Lloyd et al., 2019). With respect to the impact of soil-P losses on aquatic ecosystems, it is worth noting the distinction between natural soil-P content and anthropogenic soil-phosphorus. The former may be held within P-bearing minerals which may not be readily exposed to weathering processes and thus contribute slowly, over geological time, to the release of dissolved P in soil porewaters and aquifers (Bingham et al., 2020). Once released from mineral form, this P may be taken up by soil microbiota and incorporated into dissolved or particulate organic P forms, or adsorbed to soil particles particularly in association with iron, or is flushed to adjacent surface and groundwaters, where it may be taken up by primary producers or modified by microbial communities in both soils and waters (Bryce et al., 2016; Bingham et al., 2019; Brailsford et al., 2019). In contrast, by design, anthropogenic P additions to soils, for example in mineral fertilisers, are more readily open to weathering and dissolution processes and more immediately available for biotic uptake. Image courtesy of Oxford University Press and orginally published in Elser and Haygarth, 2020.

compounds present in soils and sediments, such as iron, aluminium, and calcium oxyhydroxides, can bind P from solution within a particulate phase (Sharpley et al., 2013). Soils with high P binding capacities will, therefore, capture more dissolved P from soil porewaters and slow its delivery from source to water (European Environment Agency, 2005).

Plant species with different root types and their interactions with soil biota to form below-ground networks have a great influence on water infiltration, and P mobilisation (Shen et al., 2013). The topographic gradient, vegetation cover, water balance and physical distance to the waterbody also control the rate of P delivery to rivers. For example, in temperate climates, P mobility tends to be lower in summer due to lower rainfall and subsequent water flow, whereas in autumn and winter, when more rainfall occurs, greater P delivery to aquatic systems is observed (White and Hammond, 2006; Johnes et al., 2007). The timing of rain events relative to fertiliser application is also critical. The largest cyanobacteria blooms in the Great Lakes, in the USA, occur when spring rains directly follow P application (Gildow et al., 2016). Conversely, in tropical soils in many equatorial climates, dry spells can reduce soil moisture and cause cracks and fissures, increasing sediment delivery from sporadic rainfall via overland and fissure flow through the soil (Domínguez et al., 2004; Shipitalo, 2004; Sade et al., 2010).

5.4 Future global drivers of the phosphorus cycle and ecosystem impacts

5.4.1 Population growth drives phosphorus pollution

Population growth and economic development whilst increasing water demand (Alcamo et al., 2003; Oki and Kanae, 2006) will – without intervention - simultaneously cause greater water pollution (Vörösmarty, 2000; Heathwaite, 2010), effectively reducing the availability of clean water. Together they are considered to represent the dominant stressors on global freshwater (Vörösmarty, 2000; Kummu et al., 2010). Whilst 71% of the world surface is covered by water, only 2.5% of this is fresh water (Oki and Kanae, 2006). Most of this is not freely available or renewable as it is stored in glaciers or reserves of paleo-water (aquifers formed during wetter periods in recent geologic history, which are effectively non-renewable) (Oki and Kanae, 2006). Almost 40% of the earth's available freshwater resources are used for agriculture, industrial or domestic services, and will be polluted through these practices (Oki and Kanae, 2006; Schwarzenbach et al., 2006). This leaves an estimated 60% to support freshwater ecosystems and the remaining services (Millenium Ecosystem Assessment, 2003) they provide.

Population growth and economic development over the next decades are expected to dictate the relationship between water supply and demand to a much greater extent than climate change (Vörösmarty, 2000), although clearly, the combined effects of all pressures will be considerable. A key consideration is the link between water quantity and quality, where, for example, the effects of nutrient pollution may be most disruptive when the water supply is low and demand high. Under modelled projections, a substantial increase in relative water demand can be expected (OECD, 2012). Currently, half a billion people face severe water scarcity all year round, with nearly two-thirds of the global population experiencing severe water scarcity for one month a year (Mekonnen and Hoekstra, 2016). Although arid climates will face the challenge of water shortage, in water-rich areas the challenge will not be providing adequate quantities of water, but providing supplies that are of sufficient quality for use (Vörösmarty, 2000; Heathwaite, 2010).

5.4.2 Climate change and phosphorus transport from land to sea

The temperature of lakes, oceans and the atmosphere is rising, atmospheric concentrations of greenhouse gases have increased, snow and ice have diminished, and sea levels have risen (IPCC, 2018). The spatial and seasonal distribution of fresh water will change under the pressures of a changing climate (Oki and Kanae, 2006; Şen, 2009). Understanding how these changes will affect the quality and ecology of freshwater and coastal ecosystems at a global and regional scale is complex due to the variation in the geographical, hydrological, and climatic systems involved (Vörösmarty, 2000; Oki and Kanae, 2006; Rabalais et al., 2009; Şen, 2009; Woodward et al., 2010). Further, interactions between climate change effects on nutrient delivery and increased nutrient input to agricultural systems associated with intensification are unclear but are likely to exacerbate P pollution (Forber et al., 2018). We outline some of these complex interactions and their impacts on P delivery but stress that a comprehensive global-scale analysis of the effects of climate change on P delivery to freshwater and coastal ecosystems is beyond the scope of this chapter.

An increase in atmospheric temperature may cause an increase in precipitation intensity and alter rainfall patterns regionally (Harper et al., 2005; Alcamo et al., 2007; Şen, 2009). An increase between 1.5 and 4.5 °C in global temperature is predicted to increase global mean precipitation by 3 to 15% (Şen, 2009). Precipitation is expected to increase in higher latitude regions and some areas of the tropics and decrease in sub-tropical regions in the coming century (IPCC, 2019), resulting in expected significant changes in rates of P transfer from land to water (Ockenden et al., 2017). Increased rainfall may have dual effects: in the short-term, it can reduce nutrient concentrations and algal blooms in lakes due to greater flushing of the system (shorter hydraulic residence time) but in the long-term, it can stimulate blooms due to increased nutrients associated with runoff (Paerl et al., 2020). In arid and semiarid landscapes, a decrease in precipitation and an increase in the occurrence of droughts are expected (Alcamo et al., 2007; IPCC, 2018). The drying of soils may increase P transport via erosion following heavy rain. Increased air temperatures in some regions will increase evapotranspiration and may cause a reduction in surface runoff (Şen, 2009). A decrease in surface water flow may reduce dilution of point source pollution in waters, so increasing nutrient concentrations (Paerl and Huisman, 2009; Rabalais et al., 2009; Şen, 2009), but may also reduce the mobility of soil-bound P, and thus reduce diffuse pollution to proximal waters. Increased rainfall will result in greater nutrient delivery to coastal zones, potentially enhancing eutrophication and hypoxia (i.e. low or depleted oxygen in a waterbody) (Rabalais et al., 2009; Sinha et al., 2017). An expected rise in sea level caused by anthropogenic warming may increase P inputs to coastal areas due to the exposure of more land for erosion, the loss of natural buffers such as wetlands and mangroves, and P mobilisation through greater soil water saturation (IPCC, 2019). The interactions between these drivers are complex and make predicting the overall impact of climate change on nutrient transport difficult.

5.4.3 Multiple stressors and ecological responses

The effects of climate change are likely to modify ecological responses to P enrichment within lakes and coastal zones (Figure 5.6; Moss, 2010).

In modelled scenarios, the change in annual mean surface water concentrations of P in three oligotrophic lakes in New Zealand by 2100 under predicted temperature rise (IPCC-A2 scenario) would be equivalent to increasing P loading by 25 to 50% (Trolle et al., 2011).

Warming at an average rate of 0.34 °C per decade is already occurring at a global scale as indicated in increasing surface

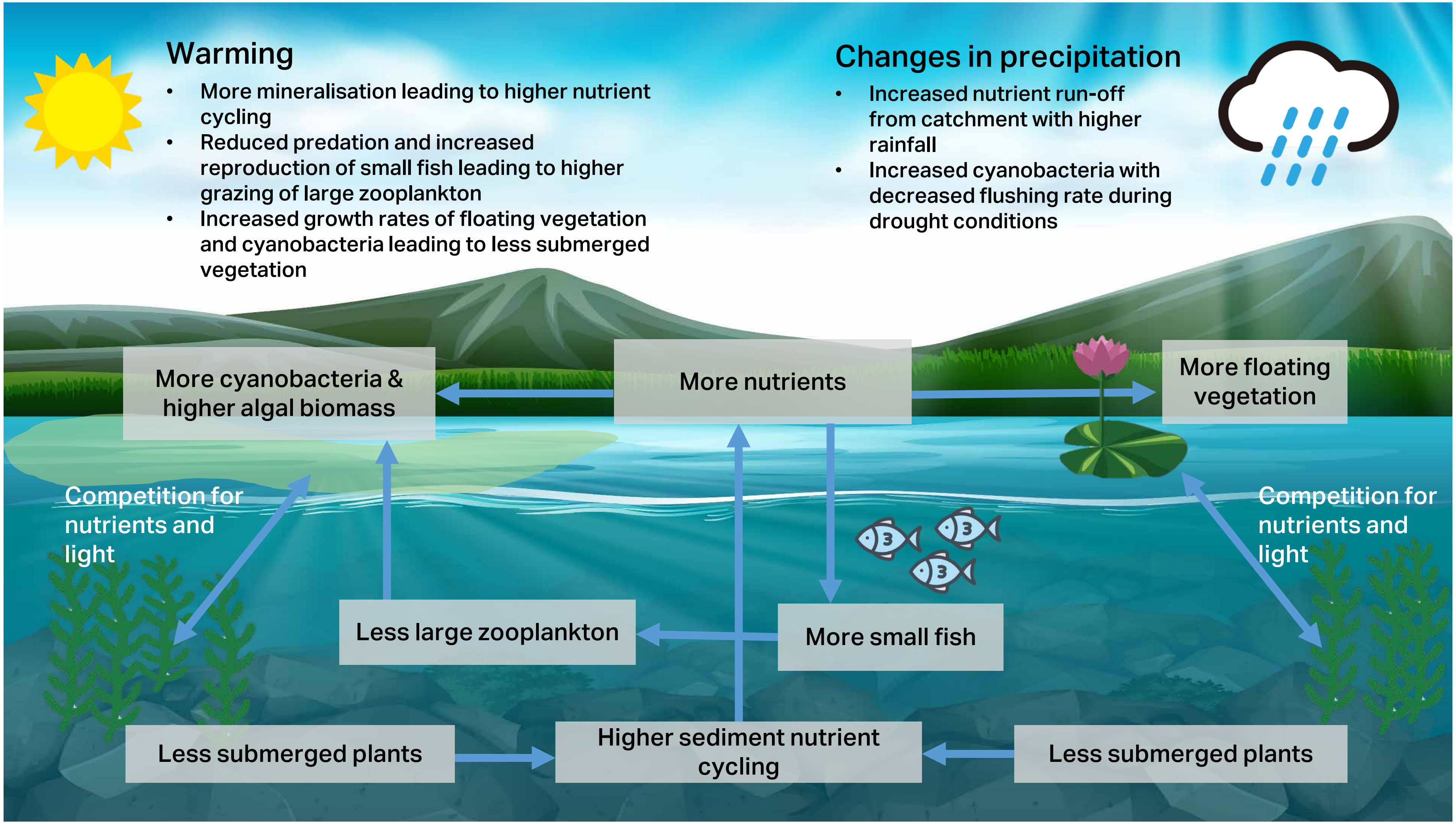

Figure 5.6 Climate change is expected to alter many components of aquatic ecosystems, modifying their responses to nutrient loading. This figure, adapted from Moss et al. (2011), summarise the expected alterations in shallow lake systems, including changes to nutrient loading, concluding that the cumulative impacts of climate change will result in a loss of biodiversity to favour greater dominance of cyanobacteria in lakes. Images sourced from https://www.vecteezy.com/.

water temperatures in lakes (O'Reilly et al., 2015). Increasing temperatures result in longer and stronger lake stratification, leading to more hypoxic conditions which in turn releases more iron (Fe) bound P from bed sediments, as Fe^{3+} is reduced to Fe^{2+} releasing the bound P (Steinman and Spears, 2020). Although the global impact of these climate change effects on water quality extremes is not yet fully understood (Michalak, 2016), some authors propose that they will increase the global extent, frequency, and intensity of cyanobacteria blooms in freshwater and coastal ecosystems (Paerl and Paul, 2012; Xiao et al., 2019). Evidence from large-scale empirical studies is available to support this view. For example, in an analysis of 494 central and northern European lakes, cyanobacterial abundance increased in most lake types with warming and decreased with extreme precipitation events (Richardson et al., 2018). Oxygen depletion is expected to increase in both freshwater and coastal ecosystems as a result of the cumulative effects of increased nutrient loading, stronger stratification, and higher water temperatures (Rabalais et al., 2009; Jeppesen et al., 2014), although an increase in the severity of storms may partly disrupt hypoxia, at least in tropical coastal ecosystems (Diaz and Rosenberg, 2008; Rabalais et al., 2009). In general, the abundance of large fish is expected to decline in lakes in favour of smaller rapidly reproducing fish under warm and eutrophic conditions (Moss et al., 2011), potentially reducing fishery performance, although local conditions will likely moderate this response. A comprehensive assessment of regional manifestations of climate change and nutrient enrichment is vital to inform novel strategies to address future ecological degradation at a global scale, including climate change resilience planning and the setting of appropriate nutrient reduction targets.

Global sustainable P strategies to control eutrophication should consider other stressors also operating at the global scale. This is well encapsulated in the planetary boundaries concept which identifies nine processes and systems that collectively regulate the resilience of the Earth System (Carpenter and Bennett, 2011; Steffen et al., 2015). Of these, four have already been exceeded: climate change, the integrity of the biosphere, land-system change, and the disruption of the global biogeochemical cycles for both P and N. The reverse is also true, that projected changes in land management in response to climate change, including changes to fertiliser application rates and timing and a move towards climate-resilient crops, should also consider impacts on P losses from land to water (Forber et al., 2018). Such interactions will be regionally specific (Painting et al., 2013; Ockenden et al., 2017), and may result in a greater ecological and economic burden regionally (Davis et al., 2009; Harris and Graham, 2017).

We argue that to meet the growing global demands for clean water and food, we should first meet the overarching goal of delivering more sustainable P management. This should be framed within the context of scale-appropriate interventions that have an additive impact towards global-scale ambitions. Next, we introduce the key challenges and solutions associated with this overarching goal.

5.5 Challenges

Challenge 5.1: Phosphorus pollution is increasing globally

Over the course of the 20th-century, phosphorus losses from land to fresh waters almost doubled because of human activity. Whilst sources of phosphorus pollution vary between regions, they are dominated by agricultural (e.g. livestock manures and fertilisers) and wastewater discharges. In many regions, phosphorus losses continue to increase.

Phosphorus losses from land to freshwater nearly doubled in the 20th century from 5.0 to 9.0 Mt P year^{-1} while N loading increased from 34 to 64 Mt N year^{-1} (Beusen et al., 2016). Population growth and economic development have significantly contributed to this increase, through increasing demands for agricultural production of animal products and producing more P wastes. Whilst P sources associated with losses from agriculture represent the dominant anthropogenic source globally, wastewaters are the main source of P to waterbodies in some countries. However, regional variation in terrestrial P sources and the transport of P from land to sea is observed (Beusen et al., 2016). In contrast to N and carbon, the P cycle is largely decoupled from atmospheric pathways, with some localised atmospheric P deposition in areas with significant wind erosion of P-rich soils (Tipping et al., 2014) and also from coal-burning power plants (Winter et al., 2002).

Phosphorus losses from wastewater and agricultural discharges are increasing globally. In the last century, losses of P from agriculture to surface waters reached about 34% of global fertiliser use (5 Mt P year^{-1}) representing 56% of all inputs to surface waters from the land. In contrast, loading from natural sources (i.e. soils that had not received anthropogenic inputs of P) remained stable at about 3 Mt P year^{-1} (Beusen et al., 2016). Phosphorus inputs from point sources to surface waters have increased by about 500% to 1 Mt P year^{-1}. However, we note that the parameterisation of P flux models at the global scale carries significant uncertainty, as acknowledged and explained by most authors who publish on this topic. For example, Mekonnen and Hoekstra (2018) (Figure 5.7) estimated that, for the period 2002-2010, the domestic sector accounted for 54%, agriculture 38% and industry 8% of the total global anthropogenic P load to fresh waters, but other models give varying estimates of these fluxes, and locally the dominant contributing sources could come from any one of these sectors. Beusen et al., (2016) estimate the global anthropogenic P load to waters at 6.2 Mt P year^{-1} in 2000, while Chen and Graedel (2016) estimate this load at 14 Mt P year^{-1} in 2013 (not including P losses during mining).

The delivery of P through fresh to coastal waters is being reduced because of large engineering structures for energy and water supply (e.g. dams for hydroelectric power generation). Phosphorus is retained by aquatic ecosystems when water residence times are high, resulting in P storage within depositional bed sediments of rivers, lakes, and estuaries, or within their biomass (Prior and Johnes, 1998; Sharpley et al., 2013;

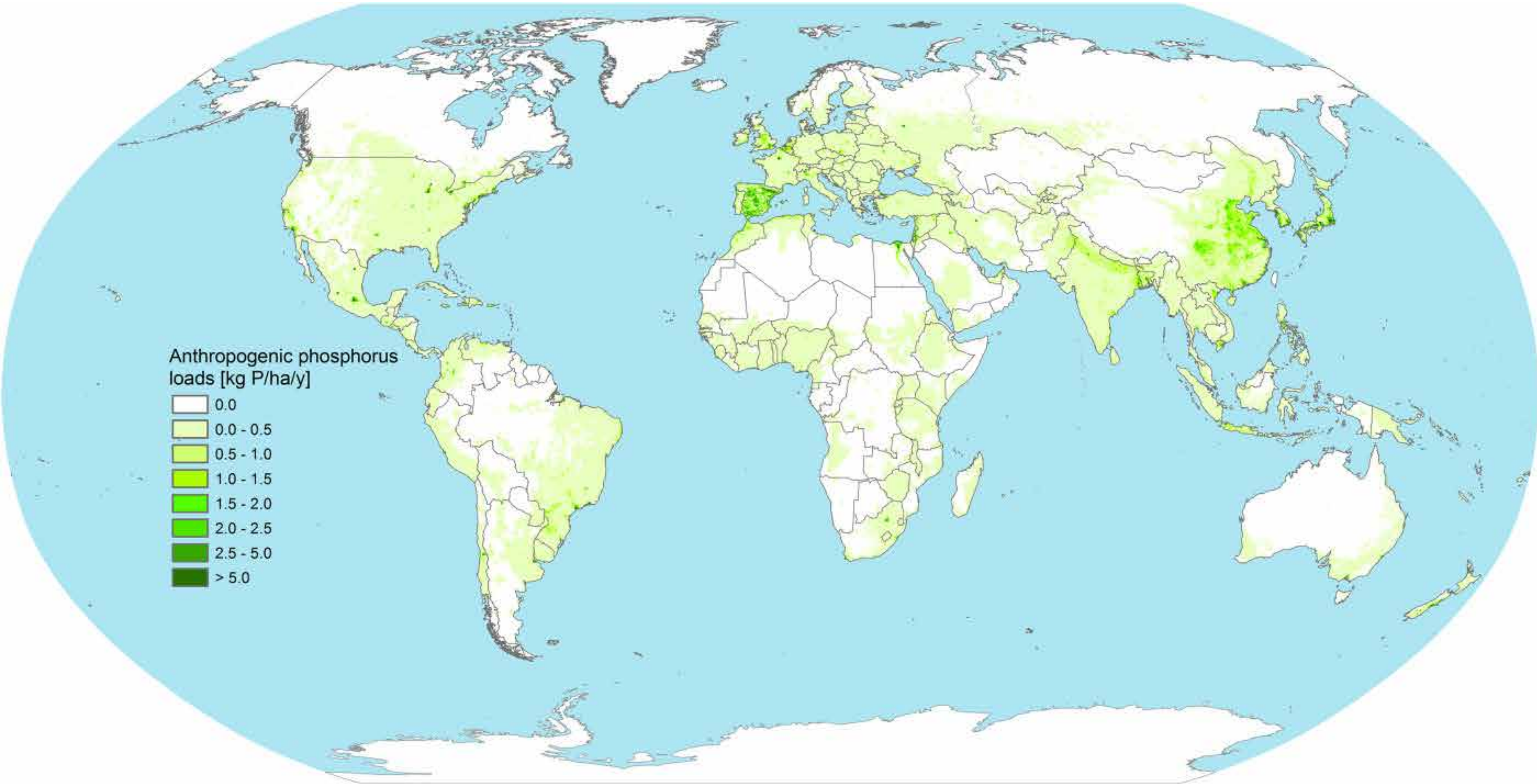

Figure 5.7 Global distribution of anthropogenic phosphorus loads to fresh water from agriculture, industrial, and domestic sectors at a 5 × 5 arc min grid for the period 2002–2010. Image courtesy of Mekonnen and Hoekstra (2018).

Johnes et al. 2020). This can alter nutrient delivery ratios, changing the ecological structure and biogeochemical functioning of receiving waters (Maranger et al., 2018).

Dam construction is a growing concern, with more than 45,000 large dams currently holding back >6,500 km^3 of water per year globally, impacting >15% of the world's river discharge (Nilsson et al., 2005), and more than 3,700 new major dams under construction or planned (Zarfl et al., 2015). The effect of reservoir creation is to slow the rate of water discharge and, also, the delivery of P from land to sea by an estimated 22% (Beusen et al., 2016).

In some regions, including parts of Africa (Nyenje et al., 2010) and India (Central Pollution Control Board, 2015), wastewaters are the dominant source of P loading, and wastes are often discharged directly to streams and rivers with no treatment. In less economically developed countries, only 8% of wastewater undergoes treatment of any kind, supporting the often-cited approximation that, globally, over 80% of all wastewater is discharged without treatment (WWAP, 2017). Phosphorus loading from point sources to water, largely from wastewater treatment discharges, can lead to high P losses across all regions with high human activity (e.g. Europe, North America, and Asia). On average, high-income countries treat about 70% of the municipal and industrial wastewater they generate; however, there is a legislative focus on P removal and not P recovery and recycling (see Chapter 7). Wastewater treatment drops to 38% in upper-middle-income countries and 28% in lower-middle-income countries; low-income countries treat about 8% as stated above. There is a clear need to increase the proportion of treated wastewater globally to reduce P and N loads.

In many countries, including China, North America, Europe and Brazil, excess mineral P fertilisers have been applied (Shen et al., 2013; Bouwman et al., 2013b; Jiao et al., 2016), leading to soil P surpluses (MacDonald et al., 2011). In such cases, soils can become super-saturated with P when binding capacity is exceeded leading to an increased risk of P delivery to proximal waters. However, we note that regions differ strongly with respect to their historical P surpluses. For example, the historical P surplus in Europe is much larger than that in North America or Brazil (Bruulsema et al., 2019). Diffuse sources related to the erosion of P-rich soils in arable and intensively grazed systems can lead to high rates of particulate P transport, especially in regions with intensive livestock production systems and/or high fertiliser application rates (Bouwman et al., 2017; Powers et al., 2019).

In cities in arid areas, there is a heightened risk of elevated P concentration in waterbodies due to low levels of wastewater dilution by rain (Nyenje et al., 2010). Further compounding these issues is the rapid expansion of cities, rural to urban migration and a significant increase in per capita generation of wastes coupled with growing water demand (Saha et al., 2010; Powers et al., 2019).

Challenge 5.2: The global impacts of phosphorus pollution are not well quantified

Elevated phosphorus concentrations in freshwater and coastal marine ecosystems are contributing to the unprecedented loss of freshwater biodiversity and the growing global phenomenon of freshwater and marine 'dead zones'. However, the true scale of the problem is difficult to estimate as baseline data are lacking across all regions and scales. Long-term monitoring programmes are necessary to track and study recovery following nutrient reduction strategies and to inform adaptive management initiatives.

In 2009, the IUCN reported that freshwater biodiversity is extremely threatened, perhaps more so than for other species, due to the rapid spread of pollution and invasive species in freshwater systems (Darwall et al., 2009), More recently, a wider range of data illustrating rates of biodiversity loss in waters have been collated and published at country, region and global scale. Over 83% of freshwater habitats in the EU were classed as in unfavourable condition in 2015, higher than any other habitat type (European Environment Agency, 2015). Freshwaters in North America are losing species at a rate of 4% per annum (Vaughn, 2010), five times faster than in terrestrial ecosystems. Over 25% of all freshwater species are currently threatened with extinction globally (Tickner et al., 2020) and freshwater fauna declined globally by 83% over the past 40 years, compared to 60% for all habitat types (WWF, 2018; Reid et al., 2019). In no other planetary domain is biodiversity declining so rapidly, despite the raft of domestic and international legislation requiring action to halt this decline.

Even though the available evidence on biodiversity loss and ecological sensitivity in aquatic ecosystems in response to nutrient pressures is compelling, global baseline data and evidence of direct cause and effect necessary to underpin regional scale P management programmes are unavailable. A similar situation exists for the prevalence of human health impacts associated with harmful algal blooms (Codd et al., 2005; Myers et al., 2013) where epidemiological data are rare, although, at least some water quality standards exist for human exposure to HABs (WHO, 1999; WHO, 2021). Thus, we rely on those examples where extensive monitoring data are available and for which the role of P in driving eutrophication has been confirmed, to inform a precautionary approach more generally. For example, in the EU, harmonised monitoring conducted under the EU Water Framework Directive has confirmed relationships between P loading and ecological quality indicators in lakes, though there are also strong relationships with N load, and ecological responses to nutrients are moderated through lake hydrology, depth, elevation, and geographical location (Phillips et al., 2008). This work has underpinned the development of ecological and chemical targets across Europe driving river basin management plans designed to help restore the ecology of degraded ecosystems, including transboundary programmes.

Similar approaches have been developed under the EU Habitats Directive, but with lower acceptable thresholds for P, and in other regions, including Australia, USA, Canada, China, New Zealand, and South Africa. However, although some standards have been developed for P in aquatic ecosystems there is a need for agreed and comparable stressor and ecological quality standards to support the United Nations Sustainable Development Goals (SDGs) 'Indicator 6.6.1 - change in the extent of water-related ecosystems over time' (UN-Water, 2016). Notably, out of the 11 global indicators to track progress towards SDG 6 on water and sanitation, this is the only indicator for which, at the global level, not enough country data were reported in 2016.

There is a need to develop an approach to provide P targets based on cause-effect relationships and baseline data to inform large-scale water quality monitoring and adaptive management programmes in all regions. Critically, we must deal with the challenge of developing the capacity for this facing much of the developing world. Equally importantly, we need to update existing policy, legislation, and enforcement mechanisms to include the wider range of stressors driving ecosystem decline in freshwaters and coastal ecosystems if the decline in aquatic biodiversity is to be halted (Ormerod et al., 2010; Smeti et al., 2019).

Long-term monitoring programmes are necessary to track and study recovery following large-scale nutrient reduction strategies and to inform adaptive management. It is critical that monitoring is conducted on sufficiently long-time scales to detect responses to measures implemented to improve water quality. If it is not, and measures are deemed to be ineffective, then we risk losing public trust in the need for reforms. Similarly, if measures focus solely on P reduction and ecosystems fail to respond to attempts to control a single stressor, we also risk the disengagement of stakeholders contributing to mitigation efforts and a similar loss of trust by the public. For example, in China, monitoring of 862 lakes indicated that P load reduction measures implemented since 2006, predominantly through improvements in sanitation and agriculture, resulted in a reduction in P concentrations in only 60% of monitored lakes (Tong et al., 2017). However, monitoring data also indicated that current conditions may not yet support ecological recovery and that the biggest initial responses were achieved in the most polluted sites.

Phosphorus is said to 'spiral' on its journey to the sea with delivery from fresh water being 'pulsed' depending on the architecture of the upstream catchment (Newbold, 1981; Ensign and Doyle, 2006). In complex catchments, P transport could take decades to centuries, and in some cases full chemical and ecological recovery may not be possible given the scale of human impact (Sharpley et al., 2013) and particularly if other stressors are driving ecosystem degradation. A review of the recovery of 89 case studies of lakes and coastal ecosystems following nutrient reduction measures reported that recovery times varied between aquatic life forms, ranging from 7 to 30 years for phytoplankton and invertebrates, and 24 years for aquatic vegetation (McCrackin et al., 2017). Utilising long-term monitoring data for 35 lakes in Europe and North America, Jeppesen et al. (2005) reported that ecological responses lagged nutrient reduction measures by 10-15 years.

For coastal ecosystems, Duarte and Krause-Jensen (2018) argue that not only have nutrient reductions been insufficient in many cases to drive ecological recovery but also that multiple pressures have emerged over the time scales of nutrient reduction to arrest the intended effects. Factors known to confound ecological recovery in freshwater and coastal ecosystems are numerous and our ability to detect them and account for them (e.g. climate change and hydrological controls) in large-scale nutrient management plans is increasing, but a full multiple stressor mitigation effort is still largely lacking (Ormerod et al., 2010; Duarte and Krause-Jensen, 2018; Birk et al., 2020). This form of evidence is essential in guiding the development of nutrient reduction strategies, and also provides a mechanistic understanding as to why some ecosystems recover quickly whereas others do not. There is, therefore, a clear need to extend conceptual and empirical understanding of the importance of multiple stressor interactions, globally, to inform long-term and adaptive management approaches that recognise the ever-changing landscape of stressors.

Challenge 5.3: Phosphorus losses and their impacts are expensive

The direct and indirect impacts of eutrophication are costly, in terms of losses of ecosystem services, clean up expenses, and losses to local economies. Phosphorus losses also represent a significant waste of resources. Global or regional assessments on the costs of eutrophication or the effectiveness of measures to reduce phosphorus losses are lacking. This severely compromises the ability to communicate the need for action with stakeholders and policymakers.

Currently, costs of addressing impacts of eutrophication are mostly paid by the taxpayer (see example in China in 2008, below; Wang et al., 2009), i.e. the public pays charges to the water industry for water treatment and P removal, or are absorbed by society through the loss of ecosystem services (Steinman et al., 2017). Thus, a pressing need is developing novel sustainable P management strategies that account for socio-economic gains as well as environmental gains generated by ecosystem recovery.

Of the few studies published, the costs of eutrophication of fresh waters in the USA was estimated at US$2.2 billion annually covering losses to industry, real-estate, and management for conservation of endangered species and drinking water supply (Dodds et al., 2009). For England and Wales in the UK, where water resources have been

been valued at £39.5 billion (ONS 2017), significant losses as a result of eutrophication of fresh waters were estimated at £75-114 million year^{-1} (US\$104 – 158 million year^{-1}) in damages and management interventions with additional policy response costs of £55 million year^{-1} (Pretty et al., 2003). These assessments mainly rely on 1990s data; these costs will likely now have increased. In economic terms, the cost of responding to algal blooms is predicted to increase as a result of climate warming. For example, in the UK warming may increase costs of response actions (after the Pretty et al. (2003) approach) from £173m (2018; US\$220 million year^{-1}) to >£400m in the next 40 years (Jones et al., 2020). The environmental damage costs associated with P pollution in England and Wales have more recently been estimated at £33 kg P year^{-1} (Zhang et al., 2017).

Single one-off algal bloom events can have significant and immediate costs for clean-up. For example, a month before the 2008 Beijing Olympic Games, an algal bloom in Qingdao Bay, where the sailing event was due to be held, closed large areas of the course. The bloom was triggered in part by P in wastewaters discharged to near-shore waters (Zhang et al., 2019). The clean-up costs associated with this single event were estimated at 593 million CNY (US\$87 million; Wang et al., 2009). The cost of the indirect economic loss to marine industries and the environment cannot be estimated because the relevant data are not yet available (Wang et al., 2009).

No global estimates of economic losses associated with eutrophication impacts or management exist in the literature. However, using estimates from catchment management case studies across China, the EU, and the USA we estimate, very cautiously, median costs for mitigation interventions at about US\$43 kg^{-1} P mitigated, including measures for the control of emissions from agricultural point and diffuse sources, combined sewer overflows, septic tanks and sewage treatment works, but excluding costs for urban diffuse pollution control which almost doubles the cost estimate using this method. Examples of measures and their relative cost-effectiveness for reducing diffuse emissions from agriculture are reviewed in Table 5.1 and elsewhere (Collins et al., 2016, 2018). The implementation of measures at a cost of US\$43 kg^{-1} P to control the additional 6.2 Mt P year^{-1} lost globally to fresh waters from anthropogenic sources (Beusen et al., 2016) would cost about US\$265 billion year^{-1}. It should be noted that these costs do not include other economic losses, so called damages, as discussed above. In addition, the costs for addressing decades of retention of anthropogenic P in lake bed sediment is not included. Estimates from the EU and USA indicate that the application of geoengineering interventions (Figure 5.8; Spears et al., 2013a; Huser et al., 2016) to control P retained within lakes would cost an additional US\$180 billion per 1.0 Mt P controlled. We note that 5.0 Mt P is currently retained in freshwater ecosystems each year (Beusen et al., 2016), although the contribution of anthropogenic emissions to this value is unclear.

Such estimates are hypothetical and do not represent a real cost that is being paid by society today, in terms of public or private financing to pay for environmental management and restoration and/or losses as estimated in the studies in the UK and USA mentioned above. In contrast, the cost of the anthropogenic losses of 6.2 Mt P year^{-1}

Figure 5.8 Trial of geoengineering in 2009 using a chemical amendment (e.g. modified zeolites and bentonites) to reduce phosphorus concentrations in the water column in Lake Okaro, New Zealand. Photograph courtesy of Andy Bruere, Bay of Plenty Regional Council, New Zealand.

to fresh water (of which 5.0 Mt is lost from agriculture) (Beusen et al., 2016), expressed per unit P in fertiliser is estimated at US$20 billion year^{-1}. 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] Significant advances are required to produce a more robust cost-benefit analysis associated with the implementation of specific mitigation measures to address a range of emissions source types across scales.

Mitigation at this scale is not, currently, a viable option either economically or politically. We note these values are used here to demonstrate the global scale of costs associated with P clean-up only. Significant advances are required to produce a more robust cost-benefit analysis associated with specific mitigation measures and their implementation across scales.

The need for preventative management using a combination of mitigation measures is recognised widely; the least expensive restoration project is the one you do not have to do in the first place. For example, the need to prevent degradation is implicit within wide-reaching policies and directives, including the EU Water Framework Directive (European Parliament, 2000) and Habitats Directive (Council of the European Communities, 1992), the Chinese "Water Ten Plan" (China Water Risk, 2015), the USA Clean Water Act (US Government,

[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.

1972), India's National River Conservation Plan (NRCP) (Greenstone and Hanna, 2014) and the New Zealand National Policy Statement for Freshwater Management 2020 (Ministry for the Environment, 2020). However, as Damania et al. (2019) discuss in detail, existing policies do not necessarily equate to effective regulation, failure to meet pollution control policy targets can be widespread, and policy targets are often too narrowly focused on individual stressors. These authors offer a realistic view that, although prevention is likely the most effective means of tackling global-scale water quality issues, as is the case for P, wide-scale implementation will be prohibitively expensive. The burden of preventative management should not be passed to future generations; it will simply grow. Consequently, there is a need for novel and wide-reaching agreements to support a stronger focus on preventative actions to safeguard relatively unpolluted ecosystems that are assessed as being at high risk of future degradation. Given the cost of such action and the need for urgency, we call for a list of priority ecosystems to be identified globally for the development of novel preventive management programmes; for example, building on the methodologies of the Global Environment Facility Transboundary Waters Assessment Programme (GEF TWAP). This assessment reported that, if current nutrient loading trends continue, a further 13 large marine ecosystems will be at increased risk of eutrophication by 2050 relative to 2000 (IOC-UNESCO and UNEP, 2016). It also highlighted challenges in predicting future trends of degradation in transboundary lakes and reservoirs as a result of limited data, environmental and ecological quality standards, mechanistic understanding of large ecosystem responses to environmental change, uncertainties in projected stressors, and a lack of transparency on governance issues (ILEC and UNEP, 2016).

Challenge 5.4: There is a lack of phosphorus policy and legislation covering water security

Phosphorus sustainability is not consistently enacted in regional policies and global action is needed to bring phosphorus enrichment of waters to the attention of policymakers. No global holistic policy on nutrient management in aquatic ecosystems exists. A key challenge is therefore enabling better integration of a sustainable phosphorus strategy across existing and emerging policy frameworks.

Phosphorus management in aquatic ecosystems is insufficiently considered within existing global policy (Cordell and White, 2015). This is not surprising since appropriate mitigation needs to reflect local conditions whilst working within a general framework, such as those embedded in the EU Water Framework Directive and others listed above. These frameworks have in common the objective of reducing nutrient loads to water from catchment sources and are generally focused on ecological improvements. Undoubtedly, evidence points clearly to the need for targeted within catchment mitigation of nutrient fluxes to waters (e.g. Lloyd et al., 2019) to achieve improvements in freshwater ecosystems, and to a lesser extent in coastal-marine ecosystems (IRP, 2021). However, no concrete global-scale target exists concerning P mitigation, specifically, though nutrient reduction ambitions have been agreed upon. For example, SDG Target 14.1 aims by 2025 to "…prevent and significantly reduce marine pollution of all kinds, in particular from land-based activities, including marine debris and nutrient pollution". The United Nations Convention on Biological Diversity (CBD; CBD, 2020) aims to "Reduce by 2030 pollution from excess nutrients, [inappropriate use of] biocides, plastic waste and other sources, [in accordance with the existing or future specific international processes] by at least [50%]". Arguably, combined N and P targets should not be inclusive as these nutrients exhibit fundamentally different behaviours, but it is necessary to manage both in tandem to control the major nutrient stressors driving ecosystem damage and to avoid 'pollution swapping' (Lloyd et al., 2019). Nevertheless, there is a clear need for evidence to support specific P reduction targets to provide consistency across existing global policies. Any proposal to alter the current global cycle of P should also acknowledge wider impacts and interactions with other key policy areas and stressors (Heathwaite et al., 2003, 2005; Sharpley et al., 2008; Buchanan et al., 2013). The challenge, therefore, is to integrate into existing and emerging policy frameworks a global sustainable P strategy whose primary aim is to relieve stress on the environment whilst supporting socio-economic gains at national to global scales (Chapter 9).

Global P footprinting has revealed large-scale displacement aligned with food and non-food production associated with trade. The complexities of the mineral P supply chain are discussed in detail in Chapter 2. An estimated 5.2 Mt year^{-1} of fertiliser P is traded internationally embodied in commodities, mainly travelling from developing economies to developed ones, such as the USA, Western Europe, and Japan (Yang et al., 2019). The P trade links are complex and dynamic, leading to geopolitical tensions and the displacement of environmental impacts (Figure 5.9; Hamilton et al., 2018). The global freshwater P footprint of traded goods and services increased between 2000 and 2011 from 0.27 to 0.31 Mt P year^{-1}, with 50% of the impact being borne by China, Eastern Europe, and Asia and the Pacific regions. Affluent countries have greater P eutrophication impacts, with every 1% increase in GDP resulting in a 1% increase in P impact (Hamilton et al., 2018). We note that estimates of P footprints are complex. A detailed description of the methods used to underpin the analysis described above and shown in Figure 5.9 is provided in Hamilton et al. (2018), with further context provided in Focus Box 8.1 - Chapter 8.

Industries operating at a global scale can play an important role in enhancing global P use efficiencies. Almost half of the world's population are reliant on fish for 20% of their animal protein intake making aquaculture a critical global industry, with a turnover of US$160.2 billion year^{-1} (FAO, 2016). Since about 2004, aquaculture has input more P to increase yield than it has extracted in fish biomass. In 2016, the net P load from aquaculture directly to aquatic ecosystems was 0.94 Mt P year^{-1}, predominantly to freshwater ecosystems in Asia which accounts for 89% of global aquaculture food fish production (China at 61.5%; Ahmed et al., 2019). The impacts of inefficient use of nutrients in aquaculture can be catastrophic, especially in areas of exceptional biodiversity, for example, in Brazilian freshwater and coastal fish farms where the practice can conflict with the Aichi Biodiversity Targets of the CBD (Lima Junior et al., 2018).

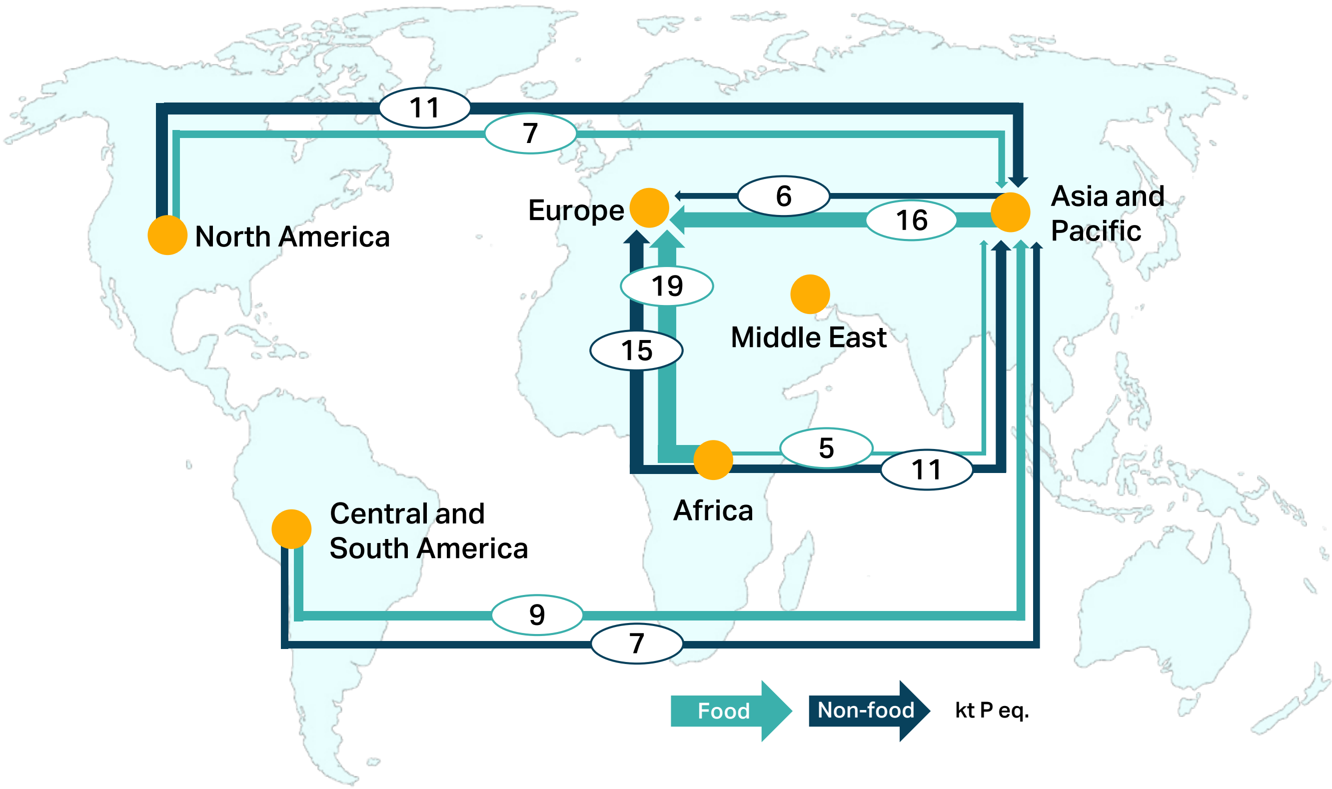

Figure 5.9 International flows of phosphorus (P) in traded food and non-food products, illustrating a complex web of P flows driven by supply and demand drivers globally. The diagram shows the top five continent-level trade-related P displacements of freshwater eutrophication (kt P equivalent year^{-1}). Arrows represent the gross flow of embodied impacts that occur in the country of origin (start of the arrow) for the consuming country (point of the arrow). Figure modified from Hamilton et al. (2018), for full methods see this reference.

5.6 Solutions

Solution 5.1: Reduce phosphorus losses and improve phosphorus use efficiency

Improved agricultural and wastewater management should be implemented to reduce losses of phosphorus from land to water. There is also a clear opportunity to improve phosphorus use efficiency in aquaculture. In order to reduce phosphorus pollution on a global scale, we must identify opportunities to decrease the amount of 'mined' phosphorus entering the anthropogenic phosphorus cycle, enhance uptake of sustainable fertiliser management approaches, and take action to close the phosphorus loop. This can be done by cutting phosphorus losses and increasing recycling and phosphorus storage within the landscape.

Reliance on mined sources of P will continue to add 'locked away' P into a global mineral cycle that is already beyond its capacity (Carpenter and Bennett, 2011; Steffen et al., 2015). By making better use of the P already circulating within the anthropogenic P cycle, through recycling (see Chapters 6 and 7) and more efficient P use within the landscape, and in the food system (see Chapter 3) we can reduce the amount of P entering the global P cycle from mined sources. Indeed, in some regions, excess mineral P fertiliser use and/or poor P management are causing 'avoidable' P losses to waterbodies (Smith and Schindler, 2009). There are also multiple opportunities to improve the efficient use of P in agricultural systems, which aim to maintain production levels with lower inputs of P fertiliser (see Chapter 4). Also, measures to reduce societal demands for P, for example, reducing consumption of animal products, where appropriate, and reducing food waste (see Chapter 8), will help to reduce the amount of P flowing within the food production system and hence cut losses to aquatic ecosystems (Withers et al., 2018). However, we highlight that in some cases, such as many countries in Africa, an increase in mineral fertiliser is required to improve soil fertility and this need should be addressed responsibly (see Chapters 2 and 3).

Opportunities are available globally to improve land-use practices towards increased P use efficiency and reduced losses to fresh and coastal waters. Shepherd et al. (2016) illustrate a conceptual model for optimisation of global P use (Figure 5.10). This model highlights the need for sustainable management across scales from global–regional–farm–plant to better balance P budgets in agricultural systems, addressing regional disconnects between crop- and livestock-dominated landscapes, and 'closing the loop' in the P cycle. Opportunities exist to adapt farming practices, for example, to harness crops that can better utilise soil residual-P stores and sources of dissolved organic P (George et al., 2018). The need for governance systems to support this model is discussed in Chapter 6 and Solution 5.4.

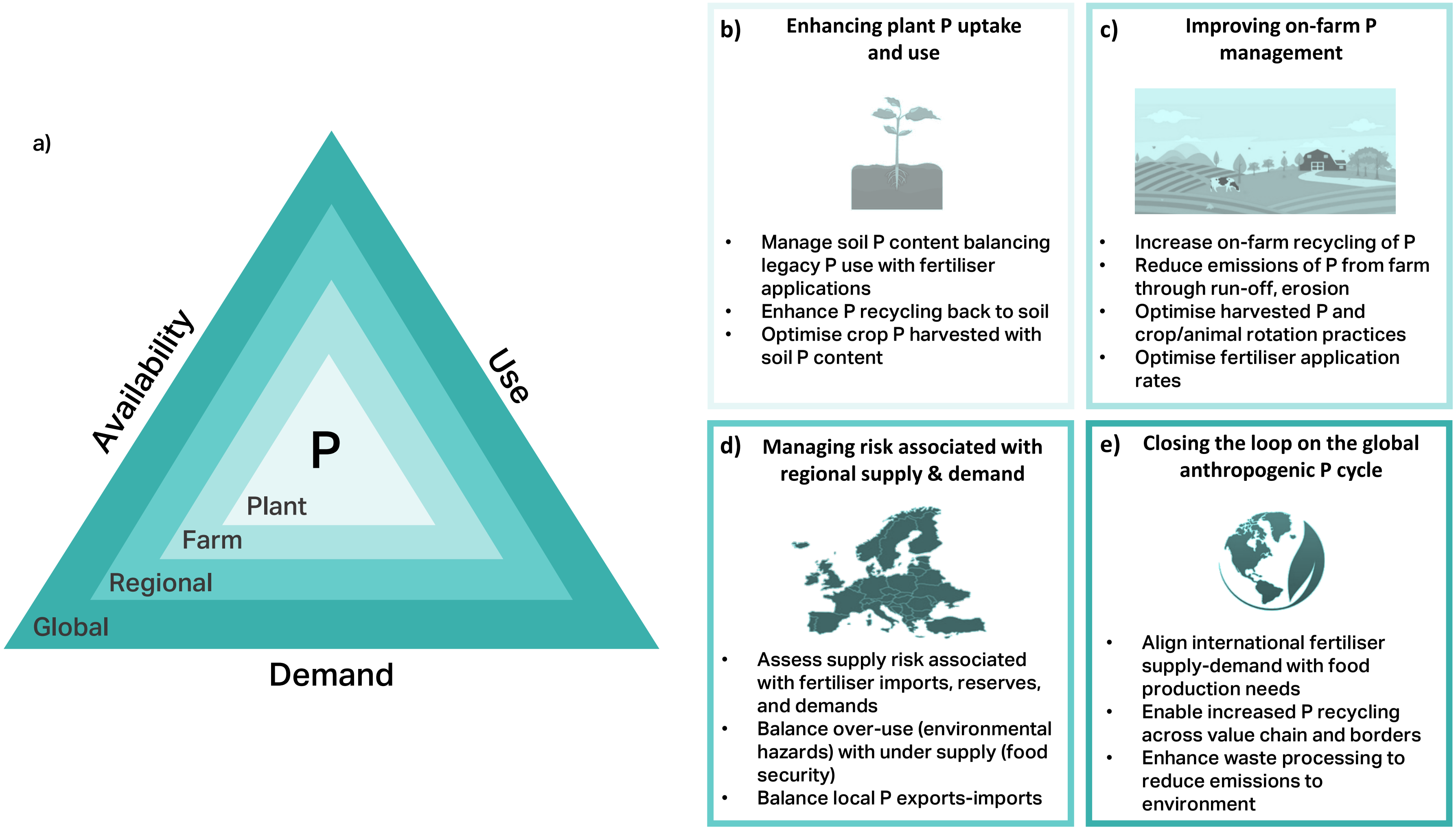

Figure 5.10 Phosphorus (P) conceptual model to optimise P use across scales to ensure (a) equitable and safe P use for all. (b) Plant P dynamics. (c) Optimising on-farm P use. (d) Regional P dynamics for regulation by governing bodies. (e) The ideal global P system of recycling and reducing losses. Image modified from Shepherd et al. (2016).

Actions to reduce losses from farms can be framed within overarching strategies trained to address national-scale farm issues (see Chapter 4). For example, the 4R Nutrient Stewardship Strategy (Right Source, Right Rate, Right Time, and Right Place; Johnston and Bruulsema, 2014) and the wider-reaching 5R approach (Re-align P inputs, Reduce P losses, Recycle P in bioresources, Recover P in wastes, and Redefine P in food systems; Withers et al., 2015a) offer global-scale application, with a recent example of its application to sugar-cane production in Brazil provided by Soltangheisi et al. (2019).

Experience in applying P loss reduction measures based on these strategies should be used to demonstrate effective farm P management initiatives in other countries or regions, especially in countries where mineral fertiliser use is increasing rapidly. This is especially important in countries where P load reductions from sewage discharges have been achieved, for example in the UK where load reductions of more than 60% (by 2020) have been achieved at a capital expenditure cost of £2.1 billion and where, despite recent reductions in fertiliser application, legacy P stocks from historical uses remain high (Environment Agency, 2019). In such cases, a balance of national-scale catchment-based planning to engage the water industry and agricultural sectors will be key to achieving further P reductions.

Common management approaches to reduce P losses from agricultural land to water are summarised in Table 5.1. To reduce P to meet ecological quality targets in any given catchment may require both increases in P use efficiency (more P applied needs to end up in crop products, so there is less to lose to waters) and control/modification of P transport pathways to intercept P before entering fresh waters. However, success in implementing these actions will rely on the effective exchange of knowledge across sectors and countries, especially smallholder farmers in low-income countries who may lack access to sufficient information and tools.

Table 5.1 Summary of efficacy and cost of diffuse phosphorus (P) mitigation strategies for different farming enterprises, compiled for land managers in New Zealand. Sources: McDowell and Nash (2012); McDowell et al. (2018); Macintosh et al. (2018).

Enterprise type	Strategy		Main targeted P form(s)	Relative effectiveness[1]	Relative cost[2]
All farming enterprises	Stream fencing	In-field management	Dissolved and Particulate	High	Low
All farming enterprises	Vegetated buffer strips		Dissolved and Particulate	High	High
All farming enterprises	Precision agriculture		Dissolved and Particulate	Very high	Low
All farming enterprises	Low water-soluble P fertiliser		Dissolved and Particulate	Medium	Low
All farming enterprises	Optimum soil test P concentration		Dissolved and Particulate	Low	Low
All farming enterprises	Refurbishing and widening flood irrigation bays		Dissolved and Particulate	Very high	High
All farming enterprises with forage crops	Restricted grazing of winter forage crops		Dissolved and Particulate	High	Medium
Cropping	Bunds to prevent runoff from leaving the field		Dissolved and Particulate	Very high	High
Cropping	Contour cultivation		Dissolved and Particulate	Very high	Low
Cropping	Cover crop		Dissolved and Particulate	Medium	High
Cropping	Minimum tillage		Particulate	High	Low
Cropping	Tillage of wheel track to improve infiltration		Dissolved and Particulate	Medium	High
Dairy	Greater effluent pond storage and deferred irrigation		Dissolved and Particulate	Medium	Low
Dairy	Low-rate effluent application to land		Dissolved and Particulate	High	Low
Red deer	Alternative wallowing		Particulate	Very high	Medium
Red deer	Preventing fence-line pacing		Particulate	Low	High

Enterprise type	Strategy		Main targeted P form(s)	Relative effectiveness[1]	Relative cost[2]
All farming enterprises	Sorbents in and near streams	Amendment	Dissolved and Particulate	Medium	Very high
All farming enterprises	Tile drain amendments		Dissolved and Particulate	Very high	Medium
All farming enterprises	Applying alum to forage cropland		Dissolved	Medium	High
All farming enterprises	Applying alum to pasture		Dissolved	Low	Very high
All farming enterprises	Red mud (bauxite) to land		Dissolved	Very high	Medium
All farming enterprises	Constructed wetlands	Edge of field	Particulate	Medium	Very high
All farming enterprises	Natural seepage wetlands		Particulate	Low	Very high
All farming enterprises	Sediment traps		Particulate	Low	Very high
Dairy	Enhanced pond systems		Dissolved	High	Very high

Prohibiting and heavily regulating against inappropriate practices can be effective, such as avoiding: the application of fertilisers, manures or slurries to soils that are water-saturated or frozen; over-stocking land with livestock; and the access of livestock to rivers for drinking water (Collins et al., 2016). Incentivisation and investment in infrastructure are also pivotal to change. For example, storage capacity and infrastructure offer farmers flexibility in terms of manure application and timing (Liu, B. et al., 2018), although future-proofing this against climatic extremes may prove difficult. A return to mixed farming systems may be effective when addressing regional disconnects between livestock-dominated and crop-dominated agriculture to better manage the 'manure-P surplus'. Nevertheless, the careful management and use of nutrients in the livestock sector remains critical to delivering water quality improvements in livestock farming catchments (Liu, X. et al., 2018; Lloyd et al., 2019; Wang et al., 2018).

In many low-income countries, addressing point sources of P, such as wastewaters and industrial P sources, represents a major opportunity. In much of the developed world control of P point sources and reductions in fertiliser applications have resulted in mixed successes in achieving large-scale nutrient load reductions leading to ecological recovery (Bouraoui and Grizzetti, 2011; Duarte and Krause-Jensen, 2018). The use of P-stripping in wastewater treatment works using chemical amendments and enhanced biological P removal has resulted in significant reductions in the P content of wastewater effluent (Morse et al., 1998; Mullan et al., 2006; Berretta and Sansalone, 2012). Now it is time to start working on complete P recycling including recovery and reuse (see Chapter 7). In China, improvements in municipal wastewater treatment have resulted in significant reductions between 2008 and 2017 in P loading to lakes relative to N (Tong et al., 2020). However, for many developing nations, basic sanitation infrastructure is absent offering limited capacity to control P losses from point sources (WHO and UNICEF, 2017).

The provision of sanitation is essential for human health. Aspirational goals to improve sanitation (e.g. SDG 6) provide an opportunity to lead global sewerage innovation by building P capture into sanitation as standard (e.g. urine-diverting toilets; Udert et al., 2016). Large-scale investments in infrastructure are required to address wastewater discharges, but these can be prohibitively expensive where the gap between operational performance and targets is large. For example, historical estimates of investment required to improve water infrastructure to meet socio-economic and Millennium Development Goals for OECD countries (the 37 member countries of The Organisation for Economic Co-operation and Development) and BRIC countries (Brazil, Russia, India and China), alone, were around US$800 billion year^{-1} (Ashley and Cashman, 2006). In many low-income countries, effective removal of pollutants and recovery of P from wastewaters is not targeted (WWAP, 2017). Improving compliance with regional standards for wastewater treatment, from both domestic (including institutions such as hospitals/schools/offices) and industrial sources, can be supported through regional and legal instruments (WWAP, 2017).

This may include the development of the 'polluter pays' principle (noting here that the 'polluter' may refer to the person who buys a product) in the context of P management to encourage behaviours that put less strain on the environment and the 'precautionary principle' where the control of hazardous substances should proceed before environmental degradation (WWAP, 2017). The domestic use of P-free detergents is currently regulated by environmental policies in the EU, the US and certain other countries (van Puijenbroek et al., 2019). Restriction on the use of P in domestic products, including detergents, should be implemented globally and extended to industrial chemicals and materials.

There is an opportunity to improve P use efficiency in key industries operating at global scales, such as aquaculture. Aquaculture is a growing concern delivering increasing inputs of P to freshwater and coastal ecosystems worldwide. More effective practices, including reducing the use of direct fertiliser application and focussing activities away from pristine ecosystems, will deliver on multiple intergovernmental biodiversity and water quality targets, including the Aichi Biodiversity Targets and many SDG targets. To achieve 'net zero P' in global aquaculture, in which P applied to enhance yield is in balance with P harvested in fish, the P use efficiency must be increased from 20% to 48% by 2050 through a range of technologically achievable industry improvements (Huang et al., 2020).

The impact of river engineering activities, both large and small, must be considered in any global P management system given they potentially capture within their biota and depositional zones one-fifth of the P transported from land to sea each year (Beusen et al., 2016). This is increasingly important as measures taken by countries to meet the Paris Agreement on climate change are leading to increased hydroelectric dam construction (Hermoso, 2017).

Solution 5.2: Implement new and utilise existing data collection systems to inform adaptive management

Monitoring programmes provide a critical link between information, evidence-based decision making, and policy development, and should be used to inform adaptive management frameworks. This is especially important given ecosystem restoration is often a long-term process, and considering the impacts on waterbodies of multiple stressors, including those associated with climate change, population growth, and urbanisation. Restoration efforts must be coupled with preventative interventions to safeguard those ecosystems that are sensitive to future increases in phosphorus input.

Monitoring programmes are critical for underpinning effective management; however, their success has been mixed (Lindenmayer and Likens, 2010). To improve this, and providing a much-needed global context, adaptive management frameworks are being developed, for example, UNEP's Framework for Freshwater Ecosystem Management (UN

Environment, 2018). These frameworks provide a vital link between information, evidence-based decision-making and policy development. Critically, they acknowledge the need for continual and long-term monitoring, given that times from land-based action and other human mitigation efforts to ecosystem responses may extend to decades. Equally important, is that monitoring includes the full range of P fractions present in waterbodies and not just those that are easily determined or assumed to be 'bioavailable'. Furthermore, monitoring needs to take place at sufficiently high frequencies in space and time that capture the episodic flux of P from land to water along overland, throughflow and groundwater flow pathways in order to reduce the uncertainties associated with load estimates derived from monitoring approaches that are low frequency or do not include all P fractions (Johnes, 2007b; Lloyd et al., 2014, 2016, 2019; Bieroza and Heathwaite, 2016; Heathwaite and Bieroza, 2021).

Restoration of impacted waterbodies is notoriously difficult to achieve, and we must now acknowledge that complete ecological recovery may be impossible in some cases, and at best is a long-term process that may only be achieved if multiple stressor mitigation approaches are adopted. In the past, ecological recovery to near-natural conditions has rarely been reported in restoration case studies and may sometimes be impossible as a result of altered baseline conditions driven by other anthropogenic pressures (Bennion et al., 2011). Restoration ecology is now being re-framed to consider opportunities to enhance ecosystem services (Costanza et al., 2014), and we propose here, to establish a global database of indicators of P use efficiency.

Opportunities are available globally to develop baseline data on land use practices that improve P use efficiency and lower losses to fresh and coastal waters. The effective cross-fertilisation of emerging knowledge, experiences and technological advances allowing better detection of pressures and effectiveness of novel interventions will be key in this respect (Figure 5.10; Shepherd et al., 2016). For example, interventions to alleviate the combined effects of sediment erosion and delivery to waters, as well as past alterations to natural water flow and hydro-morphology, will be required to underpin control of P transport and processing from freshwaters to coastal zones.

The international focus should be on making available and assessing relevant data. Where these data exist but are publicly unavailable, initiatives and agreements should be established to release them to support global assessments (Van Cappellen and Maavara, 2016). Successes in this area are evident, for example, the UNEP's GEMStat data portal currently offers access to more than 7 million data entries on freshwater ecosystems across 75 countries for a range of water quality indicators including different P fractions. The Global Lake Ecological Observatory Network (GLEON) provides high-frequency sensor data from continuous monitoring buoys across a worldwide network of lakes (https://gleon.org/data) and the Great Lakes Observing System (GLOS) provides satellite and point observation data for the Great Lakes region of North America (http://portal.glos.us/), making available a range of data useful to assess ecosystem health. Through the experience of such initiatives, the data infrastructure should be developed to produce a global data resource on P flows and impacts, extending from the land through freshwater and coastal-marine ecosystems.

Advances have been made in the detection of land-based nutrient pressures, including the use of remote sensing technologies delivering earth observation data which offer the first comprehensive global-scale assessments of water quality, including sediment loading and cyanobacteria accumulations, in inland and coastal waters (Hansson, 2007; Bresciani et al., 2011; Olmanson et al., 2011). We are now better equipped through these and other novel telemetered sensor networks and bankside analyser technologies to assess the many external factors and their interactions contributing to stress on aquatic ecosystem health (Duarte and Krause-Jensen, 2018; Lloyd et al., 2019; Birk et al., 2020). The integration of earth observation and high-resolution sensor data with ecological modelling tools stands to fill conspicuous gaps in the global data sets, increasing the confidence in management decision-making, thereby delivering environmental gains far above the scale of intervention. This is especially important when considering transboundary waters and for ensuring that any land-based action to relieve stress on fresh waters translate into improvements in coastal-marine ecosystems, as set out by the UNEP International Resource Panel (IRP, 2021).

To inform effective adaptive management strategies, these data must allow for the assessment of dominant and interacting stressors acting to degrade the ecosystem. The recovery of aquatic ecosystems following P reduction can vary depending on multiple and interacting stressors. For European lakes and rivers, where data on such stressors and ecological responses are available across scales, recent analyses covering some 14 river basins and 22 cross-basin studies have confirmed that nutrients remain the dominant stressors acting to degrade lakes (Birk et al., 2020). However, in about one-third of lakes, the ecological effects of nutrients were exacerbated by secondary stressors, including warming. For rivers, the effects of nutrients were more complex and interactions with other stressors were dependent upon the stressor type (e.g. hydrological, chemical, and thermal stressors), the scale of interest, and the biological response variable considered. Practically, this study suggests that nutrient reduction remains the primary aim for lake management across Europe, but that river restoration programmes must relieve multiple stressors simultaneously to achieve ecological targets. The analytical approach developed by Birk et al. (2020) is transferable across ecosystem types and scales and can be used to confirm the capacity of nutrient and P management to drive ecological recovery. What is clear from this work is that nutrient abatement may be a powerful tool for the mitigation of other stressors, including climate change-related stress (Spears et al., 2020), although these results should first be translated across other regions where data are made available. For example, with respect to tropical systems, nutrient and weather interactions may vary as a result of seasonal weather patterns to create novel ecological responses to P when compared to temperate ecosystems, for which the majority of evidence is currently available on cause and effect (Beklioglu et al., 2010). Global-scale stressor interaction maps with catchment resolution would be a powerful tool to support international action on combined nutrient and climate change resilience planning.

There is an opportunity to utilise emerging data and models to target P management to mitigate larger scale climate change drivers. Emerging empirical evidence shows that P loading interacts with carbon sequestration and impacts other global geochemical cycles. For example, empirical models demonstrate that an increase in eutrophication driven, in part, by increasing P loading could increase methane emissions globally by up to an estimated 1.7–2.6 Pg CO_2 eq year^{-1}, the equivalent of up to 33% of annual CO_2 emissions from burning fossil fuels (Beaulieu et al., 2019). In contrast, nutrient loading to lakes has been associated with an increase in the total global carbon burial rate from 0.05 to 0.12 Pg CO_2 eq year^{-1} in the last century (Anderson et al., 2020). These processes, and others, are geographically distinct and should be considered in line with global-scale P management strategies.

Restoration efforts should be matched with preventative interventions to safeguard those ecosystems on which the effects of urbanisation and agricultural intensification are likely to increase (UNEP, 2016; Damania et al., 2019). However, data to assess the efficacy of preventative measures is limited, where major efforts have understandably focussed on producing evidence to support restorative measures and responses. Examples of preventative approaches include more stringent control on nutrients to mitigate the impending effects of climate change in lakes (Jeppesen et al., 2017; Spears et al., 2020), and the implementation of geoengineering approaches to alleviate symptoms of nutrient enrichment and climate change in lakes and coastal ecosystems (e.g. Figures 5.1 and 5.2; Conley et al., 2009; Spears et al., 2013b). The manipulation of biological communities, for example, through fishery controls, can also help to increase ecological resilience to future environmental change (Jeppesen et al., 2012). These approaches are controversial given the scale of the application needed to promote ecological responses. For example, a proposal to address eutrophication-associated anoxia in a 60,000 km^2 area of the Baltic Sea by using pumps to mix oxygen-rich deep waters has projected infrastructure costs of US$254 million, causing some to question whether the funding would be better spent on nutrient reduction measures from land (Conley, 2012). These novel approaches should be considered promising but some are still in early developmental stages and certainly do not represent a panacea for widespread eutrophication control. Further international collaboration is required to develop the evidence base to support the selection and implementation of these approaches more widely.

Solution 5.3: Implement integrated catchment management and develop algal bloom response plans

Integrated phosphorus management strategies that cross scales will be essential in achieving improved water security globally. A road map for capacity development is required to support the wider development of long-term integrated catchment management programmes focused on phosphorus. Rapid response plans are needed to manage the risk of damage to both ecosystem and human health associated with harmful algal blooms.

Integrated P management strategies that cross scales and are aligned with strategies to control other drivers of damage to ecosystems and human health will be essential in achieving improved water quality globally. Integrated nutrient management strategies at the farm to catchment scale should target improvements in water quality and ecological responses across coupled freshwater and coastal-marine systems (McDowell et al., 2018). These should be further tailored to support 'enterprise types' (e.g. crop, livestock, intensive livestock or mixed farm systems) and the water quality and, importantly, ecological targets to be achieved. Catchment characteristics, such as soil type, nutrient retention capacity, soil buffering capacity, hydrological connectivity and flow routing in permeable versus non-permeable catchments should also be considered (Kleinman et al., 2011; Greene et al., 2015; Cade-Menun et al., 2017). To encourage uptake by catchment managers, strategies to mitigate nutrient losses need to offer practicality and cost-effectiveness or be incentivised or enforced through legislation (Collins et al., 2016). The social, cultural and economic barriers that impede the uptake of mitigation measures should also be addressed (Inman et al., 2018). The cost associated with removing or remediating the effects of P loss increases with distance from the source (McDowell and Nash, 2012). Therefore, the identification of critical source areas (CSAs - areas that account for a large proportion of P loss, but only constitute a small proportion of catchment area) has become an important management tool, which is scalable and based on reducing P losses while minimising economic costs (Wood et al., 2005; Sharpley et al., 2008; Buchanan et al., 2013; McDowell et al., 2014; Thomas et al., 2016). We encourage the development of catchment management plans based on sound scientific evidence in support of robust cost-benefit analysis, as demonstrated for Lake Rotorua, New Zealand (Mueller et al., 2016), including novel landscape planning approaches (Mueller et al., 2019). Ecosystem services valuation provides an estimate, if currently considered crude, of the monetary value of ecosystems and the benefits of the effective management of nutrients (de Groot et al., 2012; Dodds et al., 2013). The advent of indicators with which to quantify the SDGs should provide a useful, quantitative, and consistent framework with which to assess the water security costs and benefits associated with more sustainable P use.

Frameworks exist to support the development of integrated catchment management programmes across scales (McDowell and Nash, 2012; Greene et al., 2015; Lloyd et al., 2019). For example, UNEP's Framework for Freshwater Ecosystem Management (UN-Environment, 2018) and the Integrated Lake Basin Management framework of the International Lake Environment Committee (ILEC) (Figure 5.11) are both scalable to the catchment scale and acknowledge the need for accurate evidence to inform decisions within coordinated governance, policy, and institutional frameworks. We can also draw on lessons learned from established policies and directives. For example, Carvalho et al. (2019) and Poikane et al. (2019) review lessons from the EU Water Framework Directive, highlighting issues of inconsistencies in the setting and assessment of nutrient criteria and ecological indicators, as well as insufficient monitoring, financing, and governance coordination that have limited the translation of the directive into ecological improvements at the large scale. These authors call for more consistency in approach to river basin management, as well as better integration of water policy into other policy domains including agriculture, urban planning, flooding, and climate change and energy policy areas.

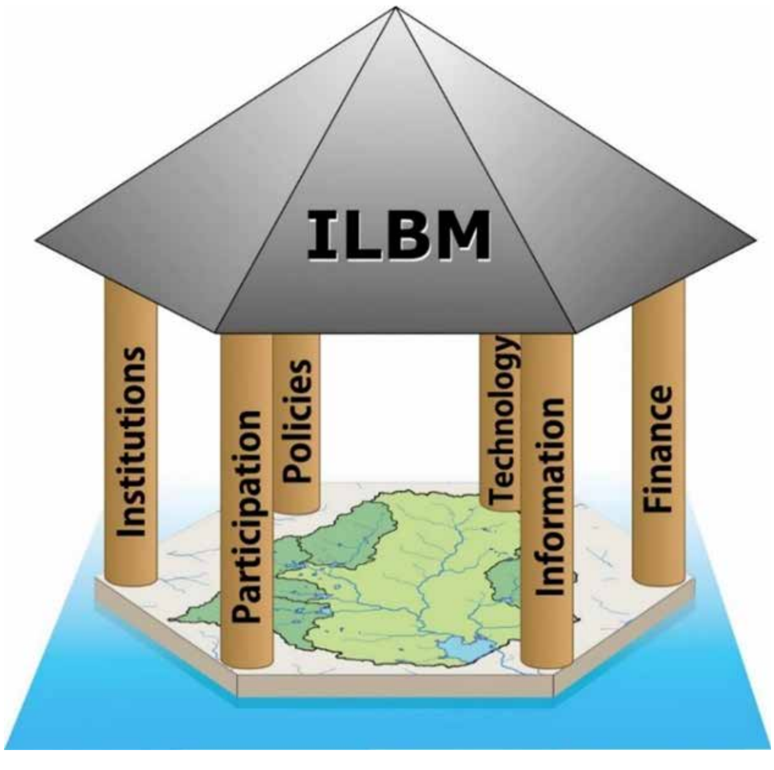

Figure 5.11 The International Lake Environment Committee's Integrated Lake Basin Management (ILBM) framework for sustainable management of lakes and their basins represents a synthesis of experiences from lake management practitioners across the world. This model is scalable to include coupled freshwater and coastal ecosystems and requires six pillars, as described below. **Institutions:** A management system with an appropriate organisational setup helps ensure sustainable benefits to watershed resource users. **Policies:** Policy tools must be better developed to facilitate concerted societal actions for sustainable watershed management. **Participation:** All stakeholders should participate in the decision-making process for sustainable management. **Technology:** Although their effects often tend to be limited in certain areas and a short period of time, physical interventions, such as shoreline and wetland restoration, provision of sewerage and industrial wastewater treatment systems, afforestation, and mitigation measures for siltation control, can play a significant role in improving the lake environment. **Information:** Scientific and public perceptions of watershed management can differ from case to case. Without knowledge generation and sharing, human and financial resources mobilised in watershed management may prove futile. **Finance:** Financial resources should come from all stakeholders benefiting from both direct and indirect use of natural resources. Efforts must be made to develop innovative approaches for generating locally usable funds. Image Courtesy of ILEC (2007).

Many case studies exist with which to exemplify the complexity of integrated catchment management across large ecosystems, including transboundary ecosystems. At this scale, political and financial constraints are commonly cited as limiting factors to success. For example, the HELCOM Baltic Sea Action Plan set out to achieve the reduction of annual nutrient loads by 15,250 t P and 135,000 t N by 2021 compared to the baseline period of 1997-2003 (HELCOM, 2007). However, the cost burden of measures in the plan has been criticised as being socio-economically inequitable and resulting in inadequate nutrient reductions. In this case, costly nutrient reductions borne by upstream countries (Russia and Poland) would provide benefits mostly to downstream countries (Finland and Sweden) (Ollikainen et al., 2019).

To address such conflicts there is a call for improved socio-economic evidence to support large-scale nutrient sustainability policies to dovetail with the scientifically advanced biophysical evidence (Ollikainen et al., 2019). This approach has been demonstrated for Lake Toba, Indonesia, where ecological impacts and increasing human health risk associated with harmful algal blooms have been driven since the 1990s by increasing nutrient pollution, associated largely with aquaculture, livestock and wastewater sectors (World Bank Group, 2018). From combining social and economic analyses with biophysical modelling of land use and P flows, based generally on the ILEC framework, 'future world' scenarios have been developed for Lake Toba to demonstrate the benefits of investments in transitioning away from existing unsustainable practices towards a more sustainable P economy, focussed on ecotourism. For Lake Toba, these benefits may include more than 3.3 million visitors by 2041 (including 265,000 foreign visitors; total income US$162 million) creating 5,000 additional jobs. Importantly, this fully costed plan offers return-on-investment estimates as well as recommendations for establishing an Integrated Lake Basin Management Platform. Furthermore, improving enforcement and regulation mechanisms to support the transition, implementing a long-term monitoring programme to inform adaptive management responses, and establishing inter-agency cooperation will help deliver the plan. Such fundamental change should be effectively managed, negotiated and communicated so that the impacts on society are limited and shared equitably. The sustainability of such large-scale transitions is uncertain, for example, where travel restrictions related to the global COVID-19 pandemic may restrict visitor numbers, in the case of ecotourism.

Businesses need to consider their exposure to changes in biodiversity, including freshwater biodiversity. The Dasgupta Review on the Economics of Biodiversity (Dasgupta, 2021), which specifically touches on nutrient impacts of aquatic ecosystems, introduces several approaches to support businesses and financial sectors in the transition towards more sustainable economic growth. These include tools with which to assess opportunities, risk and exposure of companies in relation to natural capital, biodiversity assessments, and commodities. With respect to Nature-related Financial Risk, associated with ecosystem degradation, biodiversity loss, species population decline and pollution, Dasgupta (2021) identifies three sources of risk - physical, transition, and litigation risks - building upon the terminology of the Task Force on Climate-related Financial Disclosures (2017). In the context of nutrient pollution, a physical risk may include the loss of value of real estate (e.g. Dodds et al., 2009) due to proximity to water impacted by cyanobacteria or losses to a water company or other industry (e.g. eco-tourism in Chile), associated with poor water quality (e.g. increased drinking water treatment costs). Transition risks result from adjustment towards a more sustainable economy and may arise from policy changes (e.g. the proposed decrease in aquaculture yield in the case of Lake Toba mentioned above in order to move towards a more sustainable P economy), technology changes, or shifts in market preferences or societal norms that may cause a drop in share prices or market share for companies deemed to be operating unsustainably. Finally, litigation risks may include costs borne by a company for breaching legal frameworks (e.g. fines for breaching wastewater discharge consents). Examples of such risks are provided in Dasgupta (2021) and are available in the literature. Whether or not the lessons learned from emerging climate change litigation cases will be applied to nutrient pollution of aquatic ecosystems in the future, remains to be seen. Exposure of financial institutions to risks related to resource availability and nutrient pollution of aquatic ecosystems may increase in response to unchecked nutrient pollution and climate change. For example, agricultural regulation to address biodiversity loss may lead to `stranded assets' in the agricultural supply chain, in which assets undergo unexpected devaluation or become liabilities (Caldecott et al., 2013). Guidance to support companies in identifying and planning for climate-related and other risks and opportunities through scenario analysis which extends beyond climate issues developed by the Task Force on Climate-related Financial Disclosures (2020) should be further extended to include P sustainability risks.

A road map for capacity development in the planning and implementation of integrated catchment management programmes is required with a long-term focus on reducing P and N impacts on water quality. The United Nations Development Programme approach on capacity development offers a blueprint for such a road map targeting smart institutions, visionary leadership, access to knowledge and public accountability mechanisms (UNDP, 2015). A road map for P should support harmonised approaches for monitoring, evidence-based adaptive management decision-making, and reporting to produce a global assessment of the institutional capacity to address P issues. This will be critical to target capacity development efforts where they are most needed. Importantly, this should result in the training of a new generation of integrated catchment managers focussed on developing national and transboundary P strategies and strengthening regional trust in these to deliver large-scale environmental gains (Reitzel et al., 2019). For example, effective P management may play a key role in enhancing the resilience of transboundary freshwater ecosystems in Africa to the effects of climate change.

This includes the African Rift Valley Great Lakes, upon which 50 million people are reliant for clean water and food, where climate change and eutrophication are predicted to increase concurrently in the coming decades (UNEP, 2014).

Rapid response plans are needed to manage human health risks associated with harmful algal blooms (Buratti et al., 2017), and these may be extended to ecological disasters, including mass mortalities and species extinctions. Treatment of cyanobacteria within drinking water treatment works to meet regional and national drinking water standards can be effective (He et al., 2016). However, investments may be necessary to increase treatment capacity where cyanobacteria abundance is increasing in drinking water supply reservoirs because of P and N pollution, where drinking water treatment is available. The cost associated with this action may be restrictive, especially in developing economies. On a global scale, dealing with cyanobacterial blooms, which are symptomatic of degraded lakes, has resulted in billions of dollars of new investment in water treatment plants and recurrent operational costs (Hamilton et al., 2014). At greatest risk are those communities reliant on untreated raw water. Here, measures are being developed to rapidly reduce exposure by treating the water source, for example, using low-cost chemical amendments (Douglas et al., 2016). These measures and others should be considered as short-term mitigation options for managing human health effects associated with harmful algal blooms. The four priorities for action identified in the United Nations Sendai Framework for Disaster Risk Reduction may be used to frame such plans in the context of nutrient pollution and harmful algal blooms and their consequences (UNDRR, 2015). These are first to understand disaster risk in terms of vulnerability, exposure, capacity to resist change, and potential asset losses. Secondly, to strengthen disaster risk governance to manage disaster risk at local to global scales to support the development and implementation of prevention and mitigation policies. Thirdly, to continually build resilience-to-change through investment in disaster prevention measures, including those discussed above in relation to harmful algal blooms. Lastly, to enhance disaster preparedness through constructing response and recovery plans so that asset losses are minimised in the event of a disaster and all water managers are ready and equipped to respond appropriately, including recovery and development after disasters. These steps may be supported by the large evidence base available on actions to reduce the human health risk of harmful algal blooms. For example, the WHO Water Safety Planning Approach (Jetoo et al., 2015) may be coupled with emerging monitoring techniques to provide early warning systems (Bullerjahn et al., 2016) to trigger short-term P mitigation responses.

Solution 5.4: Develop integrated policy approaches and globally coordinated phosphorus initiatives

Solutions to overcoming phosphorus inefficiencies must rely on tackling phosphorus imbalance at all scales. The development of regional targets, mandates and incentives are essential, and will often require transboundary cooperation. Where regional policies exist on phosphorus or other nutrients, experiences with these should be synthesised to inform their improvement as well as support policy development in other regions where no relevant policies exist.

Solutions to P inefficiencies should tackle P imbalance at all scales of P use and be integrated across diverse existing and emerging policy areas (Shepherd et al., 2016). Regional policies addressing sustainable P management that embrace wider regional P stewardship, and the socio-economic drivers and transitional pathways required for change towards greater P efficiency and resource protection have been proposed (Withers et al., 2015b; Jacobs et al., 2017). These approaches include the need to consider opportunities to develop a circular P economy to reduce fresh imports of inorganic P into existing farming and food processing systems (Metson et al., 2016a), recover P from different wastewaters to reduce direct effluent loadings to rivers and lakes (Bunce et al., 2018), and the potential to reduce society's total P demand by altering the P momentum of the food system through redesign, for example through dietary choice and/or genetic engineering (Childers et al., 2011; Metson et al., 2016b; Withers et al., 2018). Lowering of P surpluses would lead to lower landscape P accumulation, which would both reduce P losses to water in the longer term and increase catchment P buffering capacity, and lead to reduced emissions of greenhouse gases from surface waters. There is a need to develop policy options at a global scale to enable the integration of sustainable P ambitions across the existing policy framework (Chapter 9). From an ecosystem perspective, such options should target reductions in P emissions to freshwater and coastal ecosystems including both short-term disaster response plans alongside long-term adaptive management strategies supporting transitions towards more sustainable P economies. These should acknowledge the confounding effects of climate change on ecological degradation as well as the time scales over which the drivers of change are operating, typically, decades to centuries for global-scale nutrient cycles.

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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.

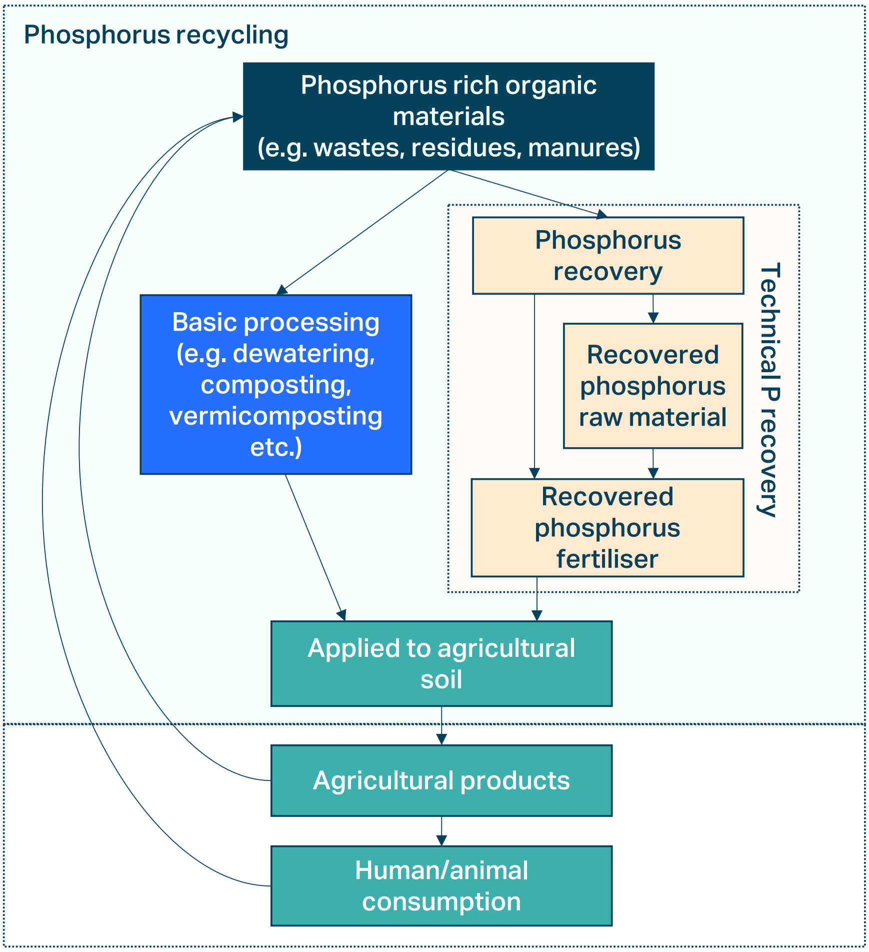

Figure 6.1 Conceptual diagram to illustrate the boundaries used in this report to distinguish between phosphorus (P) recycling and P recovery, demonstrating the circularity of P once it is in the agricultural/food system. Recycling P from organic wastes without technical P recovery is discussed in this 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.

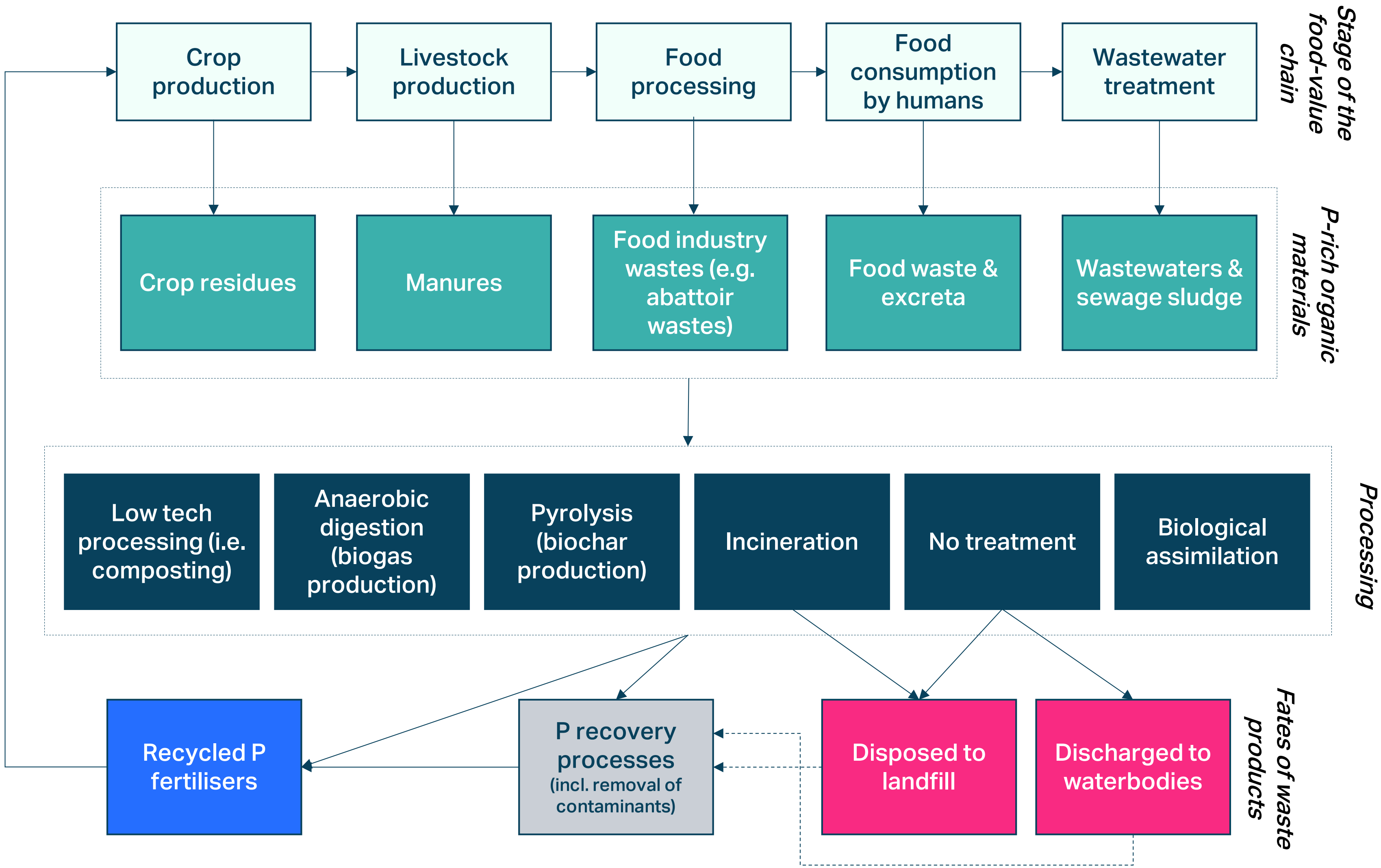

Figure 6.2 Conceptual visualisation of the food value chain showing the direction of phosphorus (P) flows (solid arrows = common flows; dotted arrows = uncommon flows), stage of the food value chain (light blue), losses of P-rich organic materials (green), common methods of processing P-rich organic 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.

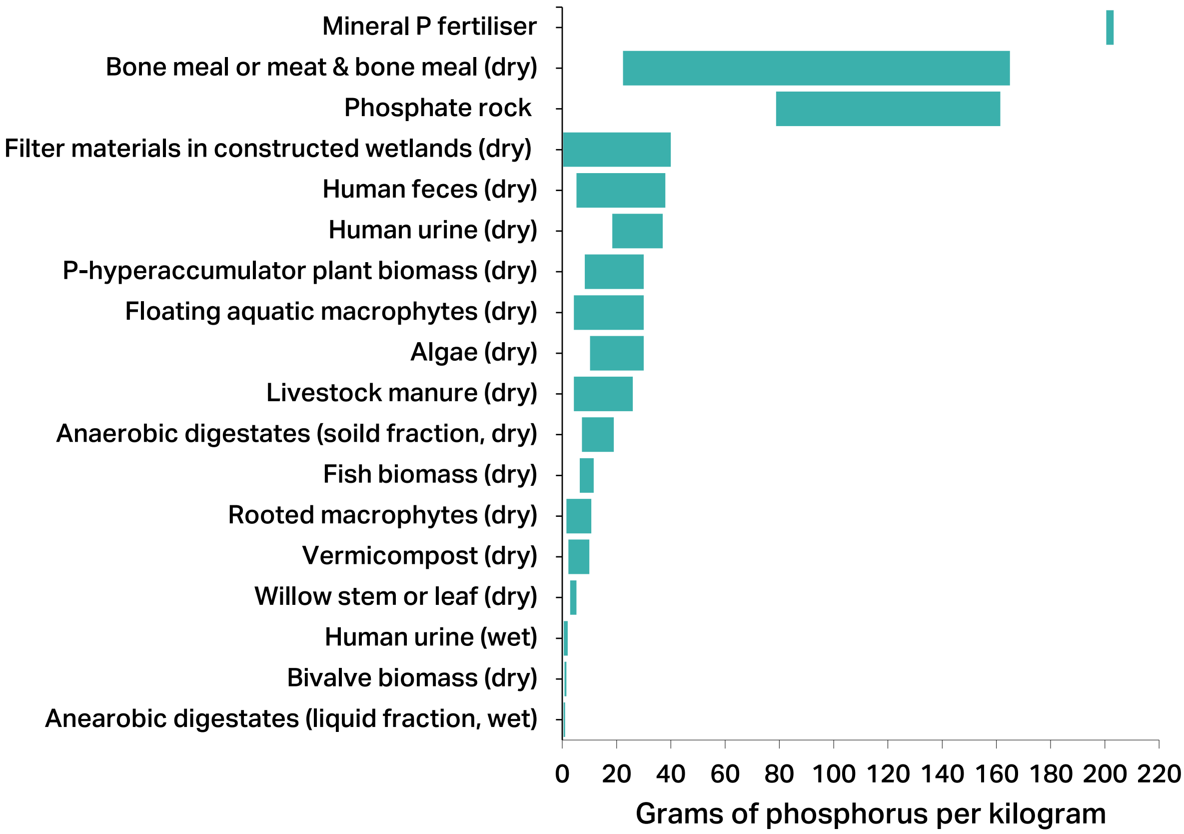

Figure 6.3 Range of phosphorus (P) concentrations (g P kg−1) of different materials, including mineral P fertiliser, phosphate rock, and several organic P materials discussed in this chapter and Chapter 7. Data source: supplementary data in Roy (2017).

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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07

Opportunities for recovering phosphorus from residue streams

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Left: A large sewage treatment site in Ukraine, showing filtration ponds. Phosphorus can be recovered from wastewaters and used to make fertilisers. Photographed by Ivan Bandura on www.unsplash.com - www.ivan.graphics

Currently large amounts of phosphorus are lost in waste streams. A global commitment to recycling nutrients in wastes and residues is needed. Phosphorus recovery provides the opportunity to recover a contaminant free, high purity source of phosphorus that can be used to create customised products, and substitute effectively for phosphorus derived from phosphate rock. Phosphorus recovery and recycling will catalyse new circular economy opportunities in line with national and international policies and directives.

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Challenge 7.1: Many waste streams and residues represent a significant untapped phosphorus resource

The phosphorus in many organic waste streams and residues, including food wastes, biosolids and abattoir residues, is commonly lost to the environment. Many phosphorus-rich wastes are managed as pollution rather than valuable phosphorus resources. The ashes of incinerated residues are often landfilled or used in building materials without recovering the phosphorus they contain. There are significant opportunities to increase phosphorus recovery in all regions.

Challenge 7.2: Recovered phosphorus materials must have a competitive commercial value

Phosphorus recovery processes that do not generate industry compatible raw materials or finished products with a clearly defined market potential may fail to contribute to phosphorus recycling. Where recovered phosphorus fertiliser match mineral phosphorus fertiliser in terms of performance, systems to support large scale production, transport and handling are currently insufficient.

Challenge 7.3: There is a lack of policy and market support for phosphorus recovery

There is a global lack of tangible policy support for phosphorus recovery, which has hindered the building of commercial markets for renewable phosphorus products, including financial instruments such as subsidies, tax incentives, or support for farmers to adopt sustainable measures. Certifying recovered phosphorus products as fertilisers can provide a significant challenge for phosphorus recovery enterprises.

Solution 7.1: Establish a global commitment to recycling nutrients in wastes and residues

Nations should commit to ambitious targets to recover and recycle nutrients from livestock manure, wastewaters, abattoir residues and industrial waste streams, whilst discontinuing landfilling phosphorus-rich ashes and their displacement into building materials. A significant increase in phosphorus use efficiency, in conjunction with good management practices to reduce and mitigate phosphorus losses is also critical.

Solution 7.2: Optimise the commercial viability of recovered phosphorus products

Phosphorus recovery technologies must produce commercially viable materials with defined market potential or that are industry compatible as a raw material for fertilisers or other products. Opportunities to produce co-value products and services (i.e. produce energy, other nutrients), and the environmental sustainability of recovery processes, should be optimised. Some recovered phosphorus products/fertilisers have a potential market opportunity to provide efficient, pollutant-free fertilisers. A key challenge for phosphorus recyclers is producing relevant volumes and homogeneous quality to meet demand. The market price of recovered phosphorus products/fertiliser alone should not define the economic feasibility of phosphorus recovery. According to the "polluters pay" principle, stakeholders could share the cost of recovery, at least in more economically developed countries.

Solution 7.3: Develop policies that support phosphorus recovery and recycling

Critical policy needs to include a regulatory framework to boost the use of recovered phosphorus materials as an alternative to phosphate rock as the primary source of phosphorus in mineral fertilisers. In some regions, the necessary infrastructure to collect wastes and residues is still required. The next step could be global binding agreements and a paradigm change: taxing the consumption of natural resources and related externalities and reducing the tax burden of renewable resources and labour.

7.1 Introduction

A significant increase in the recovery and recycling of phosphorus (P) lost in organic wastes is vital if we are to improve global P sustainability. As discussed in Chapter 6, there is great potential to recycle P (and other nutrients) by applying P-rich organic wastes and manures to agricultural soils. However, in some cases P must be recovered, detoxified, and modified, from wastes, to recycle it safely and effectively and to reach higher levels of nutrient use efficiency. In this chapter, P recovery is defined, the circumstances in which P recovery is required to support P recycling are discussed and an overview of common P recovery technologies is provided.

7.1.1 Defining phosphorus recovery and its role in phosphorus recycling

The terms P recycling and P recovery have blurred definitions in the literature (Macintosh et al, 2018). In this report, P recovery refers to processes used to isolate high-quality P from organic matter (including after an intermediate step of incineration leading to inorganic ash) into recovered products that can be recycled without further processing (e.g. struvite), or recovered P materials (e.g. calcium phosphates, phosphoric acids, white P) that can be used to make recovered P fertilisers. Fertilisers made using P recovered from wastes are also referred to as secondary fertilisers in the literature. Phosphorus recovery involves a chemical P-extraction and/or chemical bond altering process induced by reducing/increasing pH or high temperatures under oxygen-depleted conditions. Recovery usually includes a pollutant removal process resulting in a purified material or product, typically qualified for assigning an end-of-waste status, that is, the waste ceases to be waste and obtains a status of a product or a secondary raw material, at least from a legal point of view. In some literature, the definition of P recovery presented here is also referred to as advanced P recovery (Lu et al., 2016; Tonini et al., 2019). Incineration alone, while destroying most organic pollutants and removing highly volatile inorganic pollutants (mercury), is not a P recovery process (but rather a stage commonly used in the P recovery processes). Furthermore, as discussed in Chapter 6, most P-rich organic materials often undergo treatment processes before application to soils, such as dewatering, composting, vermicomposting, or anaerobic digestion, however, these are also not considered P recovery processes as it does not target a specific change in the chemical form of P, for example, extracting it from organic complexes. In this report, we define P recycling as the use of P from waste and residue streams, whether in the form of a recovered P fertiliser or organic material (e.g. manure, biosolids), to produce agricultural products. This definition is described in more detail in Chapter 6. Phosphorus recovery is, therefore, not synonymous with P recycling, but is often a stage used to process P so it can be recycled. For some P-rich organic wastes and/or circumstances, P recovery is essential to recycle the P contained in the waste stream.

7.1.2 Circumstances when phosphorus recovery is required

Phosphorus recovery provides the opportunity to recover a 'safe' (i.e. low-in or free-from contaminants), high purity source of P that can be used to create customised products, and substitute effectively for P derived from phosphate rock (PR) (Withers, 2019). In some situations, large distances can separate P-rich organic waste production in livestock-dominated areas and the croplands where they can be recycled. This is common in sites where manure or nutrient-rich urban wastes are produced in high volumes (i.e. intensive livestock production, densely populated cities) and local areas that do not contain sufficient croplands to recycle the nutrients they contain (Johansson and Kaplan, 2004; Bai et al., 2016). Transporting large volumes of bulky organic material to croplands is often not economically feasible. In these situations, P recovery processes (including solid/liquid separation) can produce recovered P materials and/or fertilisers that are cheaper and easier to store and transport.

In other situations, contaminant levels in the P-rich organic wastes and residues, even after treatment, are too high for their desired use. Processes such as composting and vermicomposting can reduce contaminants in wastes (Domínguez et al., 2004; Yadav et al., 2010; Martínez-Blanco et al., 2013) (see Chapter 6). However, pathogens, hormones, antibiotics, heavy metals, and micro-plastics can persist and can accumulate in soils/biota after repeated manure/biosolid application (Kinney et al., 2008; Hill et al., 2019). Depending on the desired use of the waste, this can pose a risk to human, animal and environmental health (Laturnus et al., 2007; Cieslik et al., 2015; Malomo et al., 2018). In some industrial applications, even trace levels of contaminants are not tolerated. Most P recovery processes produce materials that contain low to no contaminants.

A high purity sustainable P material is required by industry to make a customised product. Most customised products made using recovered P are fertilisers, however, recovered P materials can be used to manufacture a range of other products (i.e. flame retardants, feedstocks). Some recovered P fertilisers are more sustainable than mineral P fertiliser (Kraus et al., 2019; Tonini et al., 2019), but with similar P content and bioavailability allowing P inputs to soils to be carefully managed to optimise plant uptake and yield, whilst avoiding P losses to the environment.

7.2 Common processes to recover phosphorus

Selecting the most effective P recovery process depends on the type of waste treated, the resources available and the products that are required. There are more than 30 different technologies available to recover P from waste streams and new ones continue to emerge (Kabbe and Rinck-Pfeiffer, 2019). Commercially established processes of P recovery exist mainly for sewage sludge and digestate, with P recovery predominantly practised in the European Union (EU), Japan and North America (Kabbe and Rinck-Pfeiffer, 2019) (Figure 7.1).

Figure 7.1 Global distribution of P recovery from sewage installations (red = operating installations, blue = installations under construction, green = planned installations), modified from Kabbe and Rinck-Pfeiffer, (2019). In 2019, P recovery installations were mainly concentrated in only the EU, Japan and the US.

However, industrial P recovery processes have also been applied, to abattoir residues (e.g. blood, meat and bone meal), poultry litter, livestock manure, food processing wastes and industrial waste streams.

Several reviews of P recovery technologies are provided in the literature, which this document does not aim to replicate (e.g. Morse et al., 1998; Le Corre et al., 2009; Rittmann et al., 2011; Cieslik et al., 2015; Tarayre et al., 2016; Schoumans et al., 2017; Mahoo, 2018; Kabbe and Rinck-Pfeiffer, 2019; Kraus et al., 2019; Ohtake and Tsuneda, 2019; Li et al., 2019a). However, they highlight there is no 'ideal' single method to recover P from wastes, and technologies are not mutually exclusive (Walker, 2017). The number of available processes does not reflect the need to continually improve on the preceding process, but the diverse range of conditions where P recovery is required. Indeed, methods to recover P must cope with high concentrations of P and organic material in low volumes in animal, human and food wastes, through to relatively low P concentrations in large volumes of water from diffuse pollution, i.e. from runoff and erosion from agriculture (Desmidt et al., 2015).

However, whilst many processes exist, some general stages are commonly followed. The following simplified overview does not cover all technologies or processes available but rather aims to provide a conceptual overview of some of the common stages found in many P recovery processes and technologies (which may occur in different orders, combinations, and with the omission or addition of stages).

In most P recovery processes, there is an early stage to concentrate the P into a reduced solid or liquid volume by solid/liquid separation. This can be commonly achieved using iron or aluminium salts (e.g. chlorides or sulphates) to precipitate P in an insoluble metal phosphate. The metal phosphate can then be settled out by sedimentation (Morse et al., 1998). Alternatively, enhanced biological phosphorus removal (EBPR) can be used to remove P from wastewater by recirculating sewage sludge through anaerobic and aerobic conditions to optimise conditions for cell uptake by polyphosphate accumulating (micro) organisms (Oehmen et al., 2007). Volume reduction of sludge/manure/digestate often includes dewatering by solid/liquid separation. Other processes utilise anaerobic digestion, which reduces the volume of the waste and frequently enhances the dewaterability, whilst converting volatile organic compounds to biogas providing a source of renewable energy (Feng and Lin, 2017). Phosphorus can be recovered from digestates. Incineration, commonly used in the processing of abattoir residues, sewage sludge and, occasionally to poultry litter, is highly effective at concentrating P, and can result in a 90% reduction in volume and a 60% reduction in weight and destroys pathogens and degrades antibiotics (Walker, 2017). In addition, incineration can convert the chemical energy in sludge to heat and electricity with an overall positive energy balance (Adam et al., 2009). Ashes may still contain heavy metals e.g. copper and aluminium, present in the original waste/residue (Donatello et al., 2010). Iron phosphate and aluminium phosphate are not bioavailable under typical pH and redox conditions found in soils and are therefore of low value for direct use as a P fertiliser (Sartorius et al., 2012) as are ashes of incinerated wastes (Cabeza et al., 2011), as such further stages are required.

If the residue has not been incinerated, P concentration/volume reduction is often then followed by a range of physico-chemical reactions to precipitate (crystallise) or adsorb the P from the liquid fraction of the residue. Precipitation of P from wastewaters using magnesium or calcium salts is a well-established technology at a commercial scale and produces struvite ($MgNH_4PO_4{\cdot}6H_2O$), or hydroxylapatite ($Ca5(PO_4)3(OH)$), respectively (Molinos-Senante et al., 2011; Kataki et al., 2016). This process also supports wastewater treatment plant (WWTP) maintenance as struvite is a known scale deposit that can block pipes/heat exchangers, therefore recovery of struvite can lessen these impacts. Struvite can make an efficient slow-release P fertiliser and can be used as a raw material blended into mineral P fertilisers (Hall et al., 2020; Kataki et al., 2016; Li et al., 2019b). Phosphorus compounds in liquid wastes can also be recovered through adsorption onto the surface of a range of materials, including iron-based sorbents such as iron oxide particles (Kang et al., 2003). Adsorbents also include hybrid anion exchange (HAIX) or ligand exchange resins, which can combine polymer anion exchange resins with metals such as zinc (Zhu and Jyo, 2005) copper (Zhao and SenGupta, 2000) and iron (Blaney et al., 2007), to selectively remove phosphates (O'Neal and Boyer, 2015). Whilst other methods are based on the change between P adsorption/desorption of certain substances under changes in pH; e.g. zirconium desorbs P in alkali solution and is reactivated in acid solution with little deterioration in P adsorption capacity (Ebie et al., 2008). Ion exchange can also recover P from liquid wastes, and is based on undesirable ions being exchanged for solid-phase ions based on their affinity (Crittenden et al., 2005). Examples include iron-based layered double hydroxides, which utilise ion-exchange between phosphate and carbonate ions (Rittmann et al., 2011), as well as hydrated ferric oxide and copper ion loading (Pan et al., 2009; Sengupta and Pandit, 2011; Nur et al., 2014). Furthermore, capacitive deionization (developed for desalination) which creates a charged electric field can be used to accumulate ions on oppositely charged carbon electrodes (i.e. phosphates from wastewaters) (Huang et al., 2014).

Commonly following these stages, the P must then be separated/solubilised from the recovery media (i.e. adsorbents, resins, metal salts etc.), or ashes if the waste has been incinerated, to isolate the P in a usable form. For most industrial applications, including the manufacture of recovered P fertilisers, a usable form would include phosphate compounds, phosphoric acid, or white phosphorus (Huygens and Saveyn, 2018). Acid leaching can be used to recover contaminant-free, high purity phosphoric acids, from ashes and the products of physico-chemical reactions (Donatello and Cheeseman, 2013; Tarayre et al., 2016; Kabbe and Rinck-Pfeiffer, 2019). Acid leaching involves lowering the pH of the wastes to <2, commonly using sulphuric or hydrochloric acid for dissolution of P and metals (Krüger and Adam, 2015; Cohen and Enfält 2017, Cohen et al. 2019). After acid leaching, dissolved heavy metals, iron and aluminium can be separated from the dissolved P by selective precipitation, solvent extraction or ion exchange but this requires further energy or chemical input (Petzet et al., 2012). Many concentration and purification processes are technically feasible, including liquid-liquid extraction used to produce a high purity monoammonium phosphate (MAP), diammonium phosphate (DAP) and phosphoric acids (Kabbe and Rinck-Pfeiffer, 2019). Alternatively, thermochemical processes can be used to remove heavy metals (Kabbe and Rinck-Pfeiffer, 2019). Examples include sewage sludge mixed with magnesium chloride, heated to 900°C in a rotary kiln, which causes volatile heavy metals or their respective compounds to evaporate, where they can be separated by off-gas cleaning systems (Adam et al., 2009). Mixing ash with sodium compounds (e.g. sodium carbonate) and heating it to 850-900°C produces a calcined phosphate with high bioavailability but is less effective in heavy metal removal (Kabbe and Rinck-Pfeiffer, 2019).

Some common stages involved in P recovery processes, end products and advantages and disadvantages of the process are summarised in Table 7.1.

In the following sections, we draw on evidence from those regions where P recovery has been developed (e.g. North America and Europe; Figure 7.1) to highlight key challenges and solutions to mainstreaming P recovery technologies and the challenges to recycling the P they recover.

Table 7.1 Description of key phosphorus (P) recovery processes for P-rich organic materials including post-treatment products, the benefits and disadvantages of the processes, and key references providing further information.

Common starting materials	P removal method	P concentration methods (maybe used in combination)	Location of P following removal/ concentration	P recovery methods (using physico-chemical processes)	Methods to separate/ solubilise P from recovery media	Products (depending on methods)	Advantages/disadvantages
Sewage sludge, livestock manure, food wastes	None needed	Anaerobic digestion, dewatering, and solid/ liquid separation	Phosphates in solution	Chemical precipitation Adsorption	Thermal/chemical hydrolysis (optional, to enhance P recovery rate)	Struvite Calcium phosphates (HAP, DCP)	A simple process, with high purity products. Reduces struvite scaling in wastewater treatment plants. Recovery rates are low, <20% without <40% with hydrolysis. The process also requires the addition of magnesium for struvite recovery. Energy and chemical demands for thermochemical hydrolysis.
	Chemical precipitation (commonly using a metal-salt (e.g. $FeCl_3$))	Anaerobic digestion, and dewatering		Chemical precipitation Adsorption	Acid leaching (required)	Struvite Calcium phosphates (HAP, DCP)	A simple process, with high purity products, and >60% P recovery rate. The use of vivianite is limited without further processing, and magnetic separation is unproven at full/large scale
		Anaerobic digestion	Metal-P compound (e.g. $FePO_4$)	Precipitation Ion-exchange	Acid leaching (required)	Phosphoric acid Calcium phosphates (HAP, DCP)	A simple process, with high purity products, and >60% P recovery rate. The use of vivianite is limited without further processing, and magnetic separation is unproven at full/large scale

Common starting materials	P removal method	P concentration methods (maybe used in combination)	Location of P following removal/ concentration	P recovery methods (using physico-chemical processes)	Methods to separate/ solubilise P from recovery media	Products (depending on methods)	Advantages/disadvantages
Sewage sludge, animal by-products and residues	All P-removal methods	Anaerobic digestion Dewatering Incineration (required)	P in ash (solid, dry)	Magnetic separation	None needed	Vivianite	High purity "commodity" products produced, from which customised products can be produced, i.e. MAP, DAP and TSP. Produces high volumes of by-product/ waste to be handled and recycled
				Thermal P solubilisation by additives (e,g, new P compounds)	Thermal process (>850°C for calcined phosphate >1400°C for P4)	Calcined phosphate P4	Low waste in the production of calcined phosphates. P4 can be used in a range of chemical industries, including fertiliser manufacture. Medium (calcination) to high (P4) energy consumption for thermal processes, and low purity and product flexibility for calcined phosphates.

7.3 Challenges

Challenge 7.1: Many waste streams and residues represent a significant untapped phosphorus resource

The phosphorus in many organic waste streams and residues, including food wastes, biosolids and abattoir residues, is commonly lost to the environment. Many phosphorus-rich residues are managed as pollution rather than valuable phosphorus resources. The ashes of incinerated residues are often landfilled or used in building materials without recovering the phosphorus they contain. There are significant opportunities to increase phosphorus recovery in all regions.

In a global assessment of P flows in 2013, it was estimated that ~85% of the P in human excreta and other human wastes (equivalent to ~6 Mt P) were not recycled (Chen and Graedel, 2016). Whilst data is not available for the phosphorus lost in abattoir residues globally, in the EU alone ~4 Mt of animal bone biomass is produced each year (bones are extremely high in phosphorus) (Someus and Pugliese, 2018). Much of the animal bone biomass produced in the EU is incinerated and the ashes discarded to landfill (Dawson and Hilton, 2011). In an analysis of the P flows in the EU, van Dijk et al., (2016a) calculated losses from the feed and food chains amount to 1.2 Mt P year^{-1}, which is equivalent to ~50% of the annual P input to the EU food system in feed, food and fertilisers (van Dijk et al., 2016). The EU is among the regions with the highest rates of P recycling, despite this, a system from which 50% of the input is lost to waste flows can neither be considered sustainable nor efficient. Furthermore, losses from non-food production in the EU, i.e. iron ore beneficiation, pulp and paper and fibre production, add a further 0.2 Mt P year^{-1} to waste flows (van Dijk et al., 2016).

Application of manure to soils, following low technology processing to reduce contaminants (i.e. not P recovery processes) is widely practised and is discussed in detail in Chapter 6. However, in regions of intensive livestock production, manure production can be so high that regional cropland areas are not large enough to recycle the nutrients they contain. A similar situation exists for cities (especially densely populated cities), where human wastes can represent a significant source of P that may not be easily recycled due to a lack of peri-urban croplands. With a strong global trend of urbanisation, P will be increasingly concentrated in urban regions (Powers et al., 2019). Furthermore, intensive livestock production tends to cluster in locations with cost advantages, such as close to cities (Johansson and Kaplan, 2004; Bai et al., 2016) compounding this issue. Transporting large volumes of bulky P-rich organic wastes to crops, where they can be sustainably recycled, can be prohibitively expensive. In some cases, to avoid such expense, manure and wastes are mismanaged, with P losses to landfill and/or the environment (Chapter 6). In the Netherlands, livestock densities are amongst the highest globally (Backus, 2017). Animal feeds imported to maintain

Dutch intensive livestock systems, have led to manure production at levels that cause regional nutrient accumulation, with persistent harmful and degrading impacts to water quality and ecosystem health (Lürling and Mucci, 2020). Despite being unpopular with farmers, in the Netherlands policies to reduce pig manure production by incremental reduction and imposed limitations on animal production have been implemented (Schröder and Neeteson, 2008; Erisman et al., 2011). However whilst the number of pig farms in the Netherlands has decreased, from 34,000 in 1984 to 5,000 in 2015, pig production has remained relatively stable and exports have continued to grow (Backus, 2017). In such cases, alternative strategies, such as P recovery, are needed to handle the accumulation of P-rich organic wastes.

Globally, many P-rich organic wastes are managed as pollutants, rather than as a valuable nutrient resource. Whilst in many high-income countries, legislation has been developed to enforce P removal from wastewaters the driver is to prevent pollution, with little focus on recovering P in a form that can be easily recycled (Christodoulou and Stamatelatou, 2016; Jupp et al., 2020). For example, ferric dosing of wastewaters is effective at P removal but can complicate P recovery (Morse et al., 1998; Fang et al., 2005). Sewage sludge and abattoir wastes are often incinerated and the ashes disposed of to landfill or used in building materials (i.e. cement) without recovering the P they contain (Christodoulou and Stamatelatou, 2016). In an overview of legislation regarding sewage sludge management in more economically developed countries, P recovery from sewage sludge was ‘viewed as a need’, but was ‘not being carried out’, and/or was ‘yet to be developed’ in Australia, much of the EU27, New Zealand, the UK, and the USA (Christodoulou and Stamatelatou, 2016). In low-income countries, only 8% of wastewater undergoes treatment of any kind (WWAP, 2017; Chapter 5). Phosphorus losses are not just confined to organic residue streams and dairy processing waste, opportunities to recover P in industrial wastes, such as steel-making wastes, are also often ignored (Matsubae et al., 2015). Iron ore tailings and steel-making slags may contain as much as 1.0 MT P year^{-1} worldwide, equivalent to ~5% of P in world PR consumption (extrapolated from (Matsubae et al., 2015)).

Challenge 7.2: Recovered phosphorus materials must have a competitive commercial value

Phosphorus recovery processes that do not generate industry compatible raw materials or finished products with a clearly defined market potential may fail to contribute to phosphorus recycling. Where recovered phosphorus fertiliser match mineral phosphorus fertiliser in terms of performance, systems to support large scale production, transport and handling are currently insufficient.

If a P recovery process is to contribute significantly to P recycling, it should be able to generate an industry-compatible raw material (e.g. as an alternative to PR), or a finished product (i.e. a recovered P fertiliser) with a clearly defined market potential (Schipper, 2019). However, recovered P products that can replace mineral P fertilisers in terms of P concentrations and bioavailability are, currently, scarce and more costly. The most common P recovery product, struvite, is activated by plant exudates and has high bioavailability and, in comparison to mineral P fertilisers, produces similar crop yields (Hall et al., 2020). Additionally, the manufacturing systems, and transport and handling networks associated with mineral P fertiliser are well-established, global and large-scale, which gives them a commercial and economic advantage over the more costly, small-scale, and emerging P recovery technologies. Furthermore, some P recovery processes, despite achieving high P recovery rates, produce materials with low commercial viability because the physical form of the material is not compatible with existing machinery for fertiliser production. For example, struvite, when in a granular form has the physical appearance of standard, granulated fertiliser, and tends to be generally sellable, whereas struvite recovered as a sludge or fine crystals is not, as it is incompatible with fertiliser-spreading equipment and needs further processing (Schipper, 2019).

Many economic feasibility assessments of P recovery technologies are conflicting, ranging from economically unfeasible, to profitable, and focus on struvite recovery from wastewaters (Jaffer et al., 2002; Dockhorn, 2009; Cornel and Schaum, 2009; Molinos-Senante et al., 2011; Kataki et al., 2016; Kabbe and Rinck-Pfeiffer, 2019). However, it is widely acknowledged that the economic viability of P recovery is dynamic and depends on many factors (Giesen, 1999; Dockhorn, 2009; de Boer et al., 2019). That withstanding, the lack of current economic incentives to stimulate P recovery remains a significant challenge globally. The specific cost of recovering P to manufacture a recovered P fertiliser can be several times higher than the market price of mineral P fertiliser (based on equivalent weights of P) (Cornel and Schaum, 2009; Molinos-Senante et al., 2011; Mayer et al., 2016). However, context is important; local or even regional conditions and value chains can have a huge impact on the cost of P recovery processes, which may include variability in the cost for ash disposal, transportation, and uptake or competition with mineral P fertiliser industries. Upstream loading of P to wastes can impact the economic viability of P recovery. For example, the introduction of phosphate-free detergents in Dutch households reduced P levels in municipal wastewaters making P recovery from effluent less economically attractive, leading to the closure of struvite recovery plants (Giesen, 1999). The market price of PR, which spiked in 2008 (see Chapter 2 and 3), can also impact the economic potential of P recovery (Nakagawa and Ohta, 2019). Indeed it has been proposed that for P recovery and recycling from wastewater to be economically self-sufficient, PR prices need to be at least US\$100 t^{-1} (Von Horn and Sartorius, 2009). As of October 2020, prices were just above US\$80 t^{-1} (for Moroccan PR) but have been steadily declining from US\$200 t^{-1} since 2012 (IndexMundi, 2020).

However, potentially the most important determinant of economic viability for a P recovery technology is the presence of a market for its recovered P materials, products and/or recovered P fertilisers (Kabbe and Rinck-Pfeiffer, 2019; Nakagawa and Ohta, 2019; Schipper, 2019).

Challenge 7.3: There is a lack of policy and market support for phosphorus recovery

There is a global lack of tangible policy support for phosphorus recovery, which has hindered the building of commercial markets for renewable phosphorus products, including financial instruments such as subsidies, tax incentives, or support for farmers to adopt sustainable measures. Certifying recovered phosphorus products as fertilisers can provide a significant challenge for phosphorus recovery enterprises.

Policy and regulations to support P recovery and the use of recovered P products/ fertiliser are scarce or absent in large parts of the world (Cordell and White, 2015; Christodoulou and Stamatelatou, 2016; Matsubae and Webeck, 2019) (Chapter 6). With limited economic incentives for P recovery, policy and legislation are the critical drivers (Hukari et al., 2016). Whilst the type of policy support required will vary between regions, the aim should be the same; to make it increasingly easy to sell and purchase recovered P fertilisers, and increasingly difficult to apply all fertilisers (including recycled ones) in excess of crop nutritional requirements for optimal yields.

Currently, from an economic and farm systems perspective, many farmers may find it difficult to switch to the use of recovered P fertilisers even if they deliver on multiple ecosystem services (e.g. cropping systems, cover crops, buffer strips), because in general, they are more expensive to buy, and/or may require capital investments (e.g. machinery to apply P recovered fertilisers; Macintosh et al., 2019). In most regions, farmers are often not compensated for investing in more sustainable practices, including a transition from mineral P fertiliser to recovered P fertiliser use. For example, the EU Common Agricultural Policy (CAP), currently under review, does not include adequate incentives, i.e. direct payments, subsidies and tax incentives, to farmers to invest in P recovery and P recycling measures (Hermann et al., 2019). Furthermore, whilst farmers are key players in the production of raw materials, they tend to have the least power in the food-value chain, and a limited ability to demand higher food prices (to cover potential costs of more sustainable practices) (European Commission, 2015; Hukari et al., 2016; Sexton and Xia, 2018; Hermann et al., 2019; Freidberg, 2020).

In industrialised food systems, power has become increasingly concentrated in a small number of large companies (Gordon et al., 2017; Godfray et al., 2018). The concentration of power lies with a comparatively small number of retail groups, who control food retail prices, and keep most of the business value (Vorley, 2001; Clapp and Fuchs, 2009; Sexton and Xia, 2018; Freidberg, 2020). For example, in the EU28 countries, some 22 million farmers produce food for more than 500 million consumers, whilst food distribution and retail are controlled by a few large companies. In the Dutch food chain, 65,000 farmers and horticulturists, provide food to 6,500 food manufacturers, which provide food to 1,500 suppliers, which are ultimately bought by only five purchasing companies that supply 25 supermarket companies (PBL Netherlands

Environmental Assessment Agency, 2012). Furthermore, agricultural producers also face higher variability of prices for their inputs and for the products they sell, which makes their income more variable than that of other actors in the chain (European Commission, 2015).

The ongoing lack of policy and economic support has hindered the markets for recovered P materials and recovered P fertilisers. A significant challenge is achieving certification of recovered P products as fertilisers. For example, this is evident for the EU market, where certification criteria differ between nations (Hukari et al., 2016; de Boer et al., 2018). To recover P, operators must navigate market regulations, and health and environmental law. The placing of new products on the market is frequently difficult, time-consuming and sometimes even impossible due to national policies (Hukari et al., 2016; de Boer et al., 2018). In the EU this often requires attaining permits after lengthy authorisation processes for both recovery installations (e.g. environment impact assessments) and the recovered P products (End-of-Waste (EoW), REACH; Hukari et al., 2016).

All chemical substances that are traded in Europe must be approved through the European Chemical Regulation (REACH) legislative framework. Approval for struvite was obtained in 2015, alleviating an important legislative hurdle. However, an important obstacle for the reuse of recovered P products in the EU was the lack of an end-of-waste status, which is now being resolved by the EU Fertilising Products Regulation (de Boer et al., 2018). For recyclers aiming to access the EU market, implementation and interaction of the REACH and EoW criteria are central (Hukari et al., 2016). Furthermore, the legislation and regulation for recovered P and recovered P products differs between countries, which can make it challenging for companies who wish to trade beyond national markets. However, if recovered fertilisers meet the requirements of the new Fertilising Products Regulation (EU) 2019/1009 (to be fully applied in July 2022), they can be labelled as EC fertilisers (safe and effective fertilisers on the EU market including EU-wide end-of-waste). This can drastically improve the marketing position of recycled fertilisers.

7.4 Solutions

Solution 7.1: Establish a global commitment to recycling nutrients in wastes and residues

Nations should commit to ambitious targets to recover and recycle nutrients from livestock manure, wastewaters, abattoir wastes and industrial waste streams, whilst discontinuing landfilling phosphorus-rich ashes and their displacement into building materials. A significant increase in phosphorus use efficiency, in conjunction with good management practices to reduce and mitigate phosphorus losses is also critical.

Over the last couple of decades, the importance of using P-rich organic wastes as a sustainable P resource has been widely acknowledged in the literature, emphasising a need to shift the focus from P removal to P recovery in a 'usable' form, to facilitate recycling (Withers et al., 2015; Tonini et al., 2019; Smol, 2019; Jupp et al., 2020). However, to make significant improvements in sustainable P management, all countries must commit to reducing P losses in wastes and residues. This should be underpinned by clear targets to increase P recovery and P recycling, within specified time ranges.

Policy and regulation that enforce ambitious targets to recycle nutrients from wastewaters are required, globally. Cohen et al., (2019b) estimate by 2030, Europe could recover 105,000 t P year^{-1} from incinerated sewage sludge ashes, equivalent to ~10% of the P imported in mineral P fertilisers (Figure 7.2). Whilst Mihelcic et al., (2011) estimates, globally, the total P content excreted by humans (just considering available P from faeces and urine) could meet 22% of the P demand. In high-income countries, a range of P recovery processes can be retrofitted into wastewater treatment plants to recover P from wastewaters and sewage sludge (as orthophosphate or polyphosphate), or ash after sludge mono-incineration. Furthermore, current wastewater treatments can be optimised to support P recovery. For example, whilst ferric dosing is an efficient method of P removal used in wastewater treatment, the presence of iron is often perceived as negative when evaluating P recovery options, as iron may reduce the plant bioavailability of recovered P products (Morse et al., 1998; Oleszkiewicz et al., 2015; Bunce et al., 2018). However, Wilfert et al., (2015) argue a reduction in ferric dosing may not aid P recovery, as significant amounts of iron-bound P can be found in WWTP, from iron piping, equipment, and groundwater. There are ongoing discussions as to the possible value of iron phosphates as slow-release fertilisers (Chandra et al., 2009; Nieminen et al., 2011; Andelkovic et al., 2019; Wang et al., 2020), at least in iron-deficient soils. Indeed, plants and fungi use a wide variety of strategies to access iron from iron-P effectively (Bolan et al., 1987; Hinsinger, 2001; Smolders et al., 2006). Further research into P-iron interactions may help to develop methods to manipulate iron–P chemistry in wastewater treatment processes that support P recovery (Qiu et al., 2015; Wilfert et al., 2015). In many low-income countries, large parts of the population do not have access to sanitation (WWAP, 2017). Whilst P is not the priority

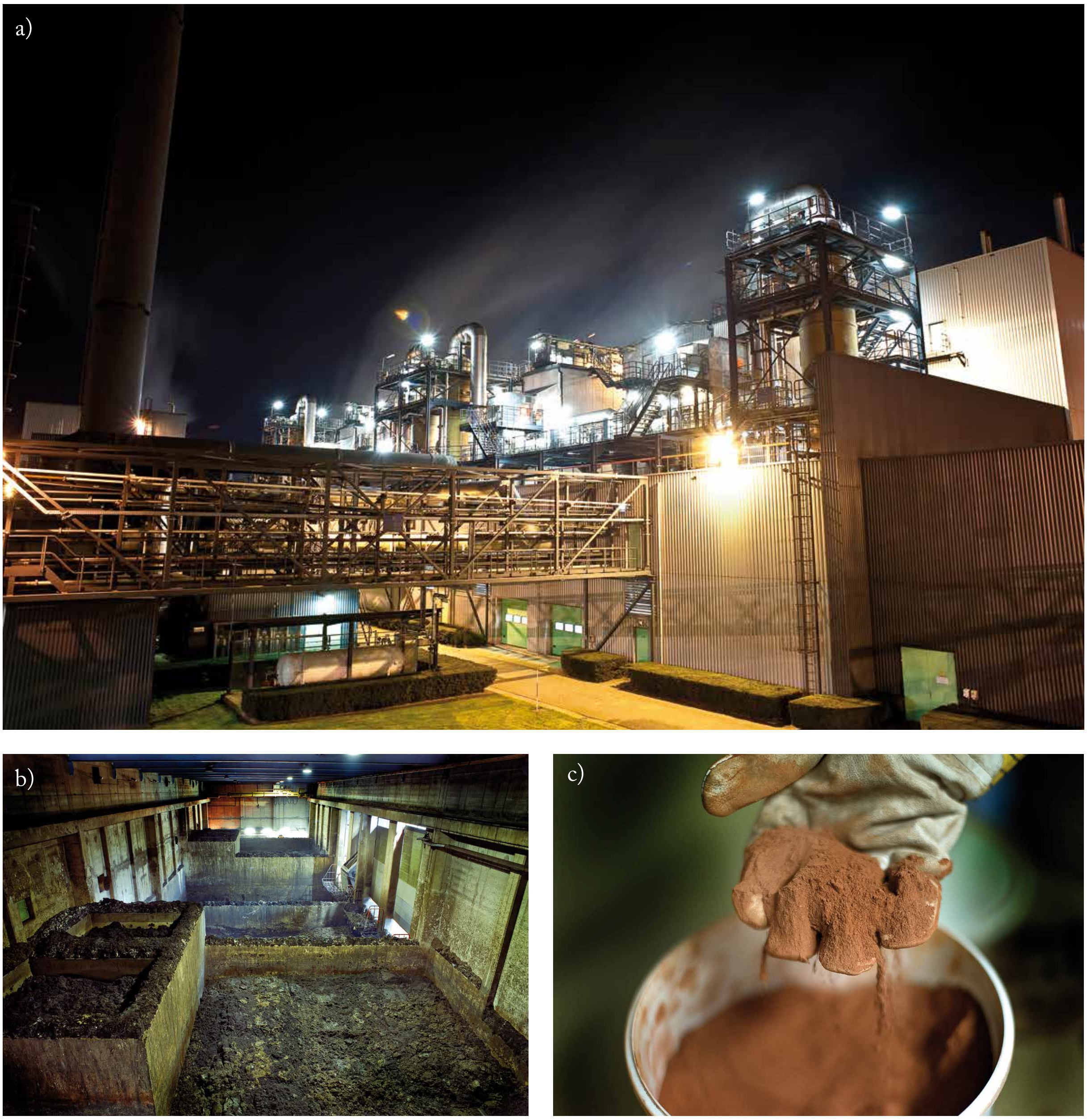

Figure 7.2 a) N.V. Slibverwerking Noord-Brabant (SNB) in Moerdijk; Europes biggest mono-incineration plant. Here, biosolids from wastewater purification in the water treatment facilities of both regional water boards and commercial parties are incinerated. **b)** Storage of ~16,000 tones of phosphorus-rich biosolids awaiting incineration and further processing. **c)** Phosphorus-rich ashes from the incineration of biosolids which can be used to make recycled fertilisers. Photographs taken by Nils Laenger - http://nilslaengner.de/

driver to improve sanitation in these regions (i.e. health risks are more important), maximising opportunities to recover P from human wastes to support sustainable agriculture is a win-win. Successful examples include the JVL Fortifer Compost Plant in Accra, Ghana, where a partnership between the local municipality, a private waste management company and the International Water Management Institute, has resulted in the production of pelletised fertiliser derived from human faecal material (IWMI, 2017).

Where manures and biosolids cannot be recycled to croplands because of large production volumes and/or transport costs, P recovery processes should be used to produce usable recovered P materials (that can be used to make recovered P fertilisers) that can be easily transported and stored. Economic value can be maximised by selecting methods to process organic materials that produce additional co-value benefits. For example, anaerobic digestion can produce renewable energy through biogas production (Mayer et al., 2016), with nutrient recovery from the digestates using struvite precipitation (Vaneeckhaute et al., 2017).

Mandatory targets to recover and recycle P from abattoir residues are required. In abattoirs, a significant loss of P is in animal bones discarded to landfill (Dawson and Hilton, 2011). Whilst the P in bonemeal has low bioavailability, it can be used as a slow-release P source (Duboc et al., 2017). For example, Thallo®, a P-rich fertiliser produced on-site at abattoirs, using a high pressure, high-temperature processing system and utilising waste products from other industries, including waste sulfuric acid and biomass power station ash (Darch et al., 2019). However, P recovery processes using heat and/or acids can recover P from bonemeal in bioavailable forms, which can be used as a replacement for PR in established fertiliser manufacturing. Furthermore, several recovered P fertilisers produced from meat and bone meal ash are already available on the market.

Opportunities to recover P lost in specific industrial waste streams such as fire extinguishers, metal surface treatment, end of life technical plastics, pharmaceuticals, electronics (Qiu et al., 2011; Ryu et al., 2012; ESPP, 2018) and steel production should also be developed (Matsubae et al., 2016). Most of these industrial streams contain very low quantities of P (compared to world PR consumption) but are concentrated and may offer feasible recovery opportunities. This may have co-benefits to industries producing the wastes, which are generally subject to stringent waste treatment and discharge requirements. On the other hand, steel-making slag is generally low in P content, 1.0 - 2.2% P (PR is around 8-15% P), but is produced in large amounts. Furthermore, dephosphorisation (i.e. processes to remove P in steel-making) can improve steel quality, however, further research is needed to identify and develop feasible/ economic methods to recover P and remove contaminants at large scale (Matsubae et al., 2011, 2016).

Landfilling of P-rich ashes (i.e. from incinerated biosolids and abattoir wastes) and their use in building materials is a waste of valuable resources and should be discontinued. This also applies to the co-incineration of P-rich organic wastes with industrial waste, and co-incineration of sewage sludge or abattoir wastes and residues in cement kilns. In both cases, P is irretrievably lost to diluted and contaminated ash or cement. The landfilling of sewage sludge is illegal in the EU and should be discontinued elsewhere. In Switzerland, P-recovery from sewage sludge will be obligatory by 2026 due to new legislation introduced in 2016. In Germany, under the German Sewage Sludge Ordinance, sewage sludge incineration ash must be stored separately for future nutrient recovery, and after 2029/32 can only be landfilled after P is recovered (Bundesanzeiger Verlag, 2017). Other countries, at least in Europe and in other high-income regions, should follow these examples.

Solution 7.2: Optimise the commercial viability of recovered phosphorus products

Phosphorus recovery technologies must produce commercially viable materials with defined market potential or that are industry compatible as a raw material for fertilisers or other products. Opportunities to produce co-value products and services (i.e. produce energy, other nutrients), and the environmental sustainability of recovery processes, should be optimised. Some recovered phosphorus products/fertilisers have a potential market opportunity to provide efficient, pollutant-free fertilisers. A key challenge for phosphorus recyclers is producing relevant volumes and homogeneous quality to meet demand. The market price of recovered phosphorus products/fertiliser alone should not define the economic feasibility of phosphorus recovery. According to the "polluters pay" principle, stakeholders could share the cost of recovery, at least in more economically developed countries.

Determining which technologies are most commercially viable, and hence should receive investment depends on region-specific factors (Cordell et al., 2011). An integrated systems framework should be used to guide decision-making for the sustainable recovery and recycling of P, as outlined by Cordell et al. (2011). This approach identifies the P that is available for recovery (i.e. quantifying P flows available for recovery from each sector), examines logistics such as regional spatial P demands (i.e. consideration for transporting products to point of use), and then identifies the tools available for P recovery, (i.e. available technologies appropriate for the region resources). Importantly, life cycle costs for P recovery, including economic, energy, environmental costs, synergies and conflicts with other industries (i.e. sanitation) are identified to ensure externalities are considered (as detailed in the analysis of Tonini et al., 2019). Through this process, the key stakeholders and institutional arrangements required to support P recovery are also identified. Failure to take a systems approach could result in investing in costly technologies that do not address the whole system, do not provide the greatest outcome for sustainability, and at worst, conflict with other related services (such as chemicals demand) (Cordell et al., 2011).

To ensure the commercial viability of recovered P products, it is important to develop P recovery processes with the direct involvement of potential users (Schipper, 2019). Common features that make recovered P materials commercially viable as an industry-compatible raw material include homogeneous quality, low to no levels of contaminants, and production levels that are high enough to ensure a reliable supply. In the best-case scenario, P from waste streams are recovered in a chemical (e.g. phosphoric acid, secondary calcium phosphates, MAP, DAP, TSP) and physical form (e.g. granules, high P content) that is already used by regional or national fertiliser manufacturers and other industries. This will allow fast and easy uptake by existing manufacturing processes. Equally, recovered P fertilisers that can be used in existing machinery and directly replace mineral P fertilisers in terms of P content and bioavailability will be more commercially viable (Schipper, 2019).

Whilst commercial viability of fertiliser is often associated with P bioavailability, standard P fertiliser tests for P bioavailability, indicated by their solubility in water or citric acid, should be reconsidered in the context of increasingly diverse recovered P fertilisers (Duboc et al., 2017). The bioavailability of many recovered P products and fertilisers is more accurately indicated by their dissolution in soil, and this can vary between soil types (Cabeza et al., 2011). Whereas fast nutrient solubility in water has been a key quality parameter of fertilisers for several decades, 'slow release', non-water-soluble P in fertilisers is increasingly being acknowledged as being important for effective nutrient supply (Shu et al., 2006; McLaughlin et al., 2011; Kataki et al., 2016; Li et al., 2019a). In the new EU Fertilising Products Regulation (EU) 2019/1009, a fertiliser is given the status of an EU fertilising product if it functions to provide nutrients to plants or mushrooms (European Parliment, 2019). However, mineral P fertilisers must fulfil certain P solubility criteria, including 40% of the declared P content must be water-soluble or 75% soluble in neutral ammonium citrate. The regulation now includes organic fertilisers and organo-mineral fertilisers, as well as other non-fertiliser products (including soil improvers, agronomic additives, plant bio-stimulants) within its scope of 'fertilising products' (Halleux, 2019). However, the high water solubility of P, a frequently used parameter for assessing the market value for mineral P fertilisers, is not justified as a good indication of bioavailability, shown by several recent comparative experiments (Cabeza et al., 2011; Duboc et al., 2017). Consequently, the plant nutrition value of some non-water-soluble recovered fertilising products may be comparable to PR-derived fertilisers and consequently should have a similar market price.

In the EU, a potential market opportunity for some recovered P fertilisers and materials is by providing pollutant-free alternatives to PR derived and non-decontaminated recycled fertilisers. The heavy metal content of municipal wastewater derived struvite is found to be significantly lower than that of PR derived phosphates (Hall et al., 2020; Forrest et al., 2008; Latifian et al., 2012) and below most regulatory limits, for example in Germany and Turkey (Antonini et al., 2012; Latifian et al., 2012; Uysal and Demir, 2013) (Figure 7.3).

The perception that the market value of recovered P products defines the economic feasibility of P recovery technologies is incorrect (Mayer et al., 2016). The market value of recovered P materials/products is among a list of the wider co-benefits of P recovery, which carry economic co-benefits (Cordell et al., 2011; Mayer et al., 2016; Tonini et al., 2019; Withers, 2019; Chrispim et al., 2019). Indeed, when comparing the externalities associated with mining PR and the manufacture of mineral P fertilisers, and those for recovered P fertilisers, the focus of P

Figure 7.3 Phosphorus recovered from wastewaters in the form of struvite produced from a Huber SE precipitation reactor. Photograph courtesy of The Sustainable Sanitation Alliance (SuSanA).

recovery processes can shift from the exclusive supply of a 'product' to a 'service' which combines decreased emissions to the environment (i.e. soil, air and water), reduction in waste generation, with the combination of high-quality P fertilisers. Societal costs incurred for recovered P products derived from sewage sludge, manure and meat and bone meal, are up to 81%, 50% and 10% lower than for PR derived superphosphate, respectively (Tonini et al., 2019). When factoring in externalities, Tonini et al., (2019) found the environmental and health life cycle impacts are often lower for P recovered fertilisers than for mineral P fertilisers, especially in areas of high livestock and population density. Furthermore, this does not factor in the risks of P depletion, or sanitation of manures, which would further modify the balance towards P recovery.

Many co-benefits remain unquantified, and therefore assessing the economic feasibility of P recovery often does not accurately represent the true net societal gains. When the total value of P recovery is accounted for, including products, services and externalities, additional incentives emerge in support of P recovery and reuse (Mayer et al., 2016; Tonini et al., 2019; Hörtenhuber et al., 2019). For example, P recovery in WWTPs is used mainly for operational benefits (i.e. reduction of struvite build-up) and is not driven by the market value of the recovered P (Kabbe and Rinck-Pfeiffer, 2019).

Phosphorus recovery technologies can be developed that carry increased value-added benefits. These may include aligning dual or multiple nutrient recovery processes alongside P recovery, such as nitrogen and micronutrients like magnesium, copper, and zinc (Timotijevic et al., 2011; de Haes et al., 2012; Kupfernagel et al., 2017). Phosphorus recovery naturally opens opportunities to recycle other nutrients, partly due to similar drivers and partly due to directing the attention of researchers and stakeholders to the related possibilities (Mayer et al., 2016; Vaneeckhaute et al., 2019; Barampouti et al., 2020). For example, currently, nitrogen in sewage sludge and wastewaters is frequently treated by nitrification/denitrification releasing nitrogen into the atmosphere, however, this can be replaced by technologies that recover both nitrogen and P, as demonstrated for biogas plants (Shi et al., 2018; Khoshnevisan et al., 2021). Anaerobic digestion can also produce renewable energy, through biogas production (Guilayn et al., 2020). The potential to produce bio-energy as a co-product, as well as optimising the use of renewable energy in the energy demand of the recovery process (i.e. for thermal treatments), can help to lower the energy footprint of P recovery technologies (Balmér, 2004; De Graaff et al., 2011). The benefits of fractionation, recovery and recycling of nutrient flows from anaerobic digestion plants are demonstrated and reported in the Horizon 2020 project SYSTEMIC (www.systemicproject.com).

Recycled P will improve farmer fertiliser security and protection against fluctuations in PR price and supply shocks. A lack of purchasing power prevents many poor farmers from accessing mineral fertiliser markets (Cordell and White, 2014). Small-scale and decentralised sanitation systems (ranging from individual onsite systems through to community-scale) have been developed due to their lower cost, or appropriateness for serving remote or low-density populations (Cordell et al., 2011). In this way, locally recovered P can contribute to farmer fertiliser security and hence food security (see Chapter 3 and 8), whilst recovered and recycled phosphates reduce the exposure of farmers and food systems to market fluctuations in PR prices. Regional factors drive the costs and prices of recycled P and are largely predictable and usually as stable as the economy in the region. Whilst recycled P on average costs more than mineral P in fertilisers, decentralisation of P recovery may lead to lower transport costs and prices may not be subject to the volatility of commodity prices (see Chapter 2).

Solution 7.3: Develop policies that support phosphorus recovery and recycling

Critical policy needs to include a regulatory framework to boost the use of recovered phosphorus materials as an alternative to phosphate rock as the primary source of phosphorus in mineral fertilisers. In some regions, the necessary infrastructure to collect wastes and residues is still required. The next step could be global binding agreements and a paradigm change: taxing the consumption of natural resources and related externalities and reducing the tax burden of renewable resources and labour.

Policy and financial support should be developed to increase the feasibility and opportunity for P recovery and recycling, this is especially important as current economic incentives are not sufficient (Hukari et al., 2016). A focus on supporting emerging industries will be key. For example, P recovery and recycling can contribute to the development of new more sustainable business opportunities. Frequently, small to medium enterprises (SMEs) provide the services associated with P recovery and recycling, potentially creating job opportunities that could reduce rural-urban migration (Steffen et al., 2015). In 2019, more than 100 P recovery plants were operational in Europe, Canada, Japan, and the US (Kabbe and Rinck-Pfeiffer, 2019). In an assessment of P flows in the EU, the P flows in effluents from livestock farming were estimated to be three times larger than the P contained in municipal waste flows (van Dijk et al., 2016) offering opportunities for P-recovery and recycling process operators in rural areas. Most suppliers and rural operators are SMEs, representing a possibility for new high-quality jobs related to agricultural activities. In addition, P recovery and recycling will catalyse new circular economy opportunities in line with national and international policies and directives. Considering global warming and finite resources, globally acknowledged by the Paris Climate Change Agreement (COP21) and the SDGs agreed in 2015, the Circular Economy is a must, with business as usual, not an option. The European Commission selected P for implementation within its "Circular economy: A zero waste programme for Europe" due to being a critical and non-replaceable element in agriculture (European Commission, 2014a). The feasibility of P-recovery within the prevailing socio-economic system could create a convincing narrative for introducing circular principles in other economic activities.

In most nations, the establishment and implementation of stringent regulations to enforce time-bound targets for P recovery (and recycling) are required. Global advocacy, and awareness-raising of the environmental benefits of P recovery and recycling, will help to improve public and political support (Matsubae and Webeck, 2019) (see Chapter 6). In the EU, the need to recover P from waste streams is already underpinned in policy through the inclusion of PR, and elementary phosphorus (P4) in the EU critical raw materials list (European Commission, 2014b). Indeed, globally, most policies and regulation regarding P recovery and recycling are currently found in the EU (Christodoulou and Stamatelatou, 2016). Currently, only Switzerland (in 2016) (The Swiss Federal Council, 2015) and Germany (in 2017) (Bundesanzeiger Verlag, 2017) have adopted regulations that make P recovery mandatory. In Switzerland, from 2016, under its Ordinance on the Avoidance and Disposal of Waste, a ten-year transition began that will make the recovery of P from sewage sludge and slaughterhouse residues obligatory (The Federal Council - Switzerland, 2016). Switzerland banned direct use of sewage sludge on land in 2006, so the regulation will lead to technical recovery and recycling in the form of inorganic products. Swiss sludge and slaughterhouse waste together represent an annual flow of 9100 t of phosphorus whereas technical recycling from the wastewater stream in Europe today totals of up to 5,000 t of P in the form of struvite (Kabbe and Rinck-Pfeiffer, 2019). A similar policy was implemented in Germany in 2018 and outlines obligatory P recovery from sewage sludge for 60% of wastewater treatment works (i.e. those that serve >50,000 people) (BMU, 2017).

Developing international targets to reduce nutrient losses that align with existing regional targets, will help to fuel momentum towards a global increase in P recovery and P recycling. The 2020 European Green Deal and with its flagship Farm-to-Fork Strategy provides an ambitious framework requiring a 50% reduction in nutrient losses by 2030, only achievable by massive improvements of full-chain nutrient use efficiency (NUE) (for definitions of full chain NUE see Chapter 5). This represents an opportunity to increase the use of recovered P fertilisers, as a sustainable alternative to mineral P fertiliser, with known and homogeneous P content allowing farmers to carefully match P inputs to crop needs (this is often difficult to achieve with manures).

Regional targets should be developed and integrated, with existing agricultural policy to ensure sufficient support is in place, for targets to be achieved. For example, the European Commission's Farm-to-Fork Strategy must be supported by the Common Agricultural Policy (CAP) and implemented by supporting policy instruments (e.g. subsidies for nutrient stewardship and biodiversity protection) in member states. In the EU within the Common Agricultural Policy (CAP), whilst currently under development the proposed 'eco-schemes' (to be implemented in 2022), may provide financial assistance to EU farmers to adopt sustainable practices (see Chapter 6). Ensuring 'eco-schemes' include the use of recovered P fertiliser as a sustainable measure is therefore important. Many less economically developed countries lack relevant environmental regulations to support P recovery, whilst in some countries/regions, significant investment in the necessary infrastructure to collect and treat P-rich waste streams is still required (Matsubae and Webeck, 2019). Subsidies and tax incentives to farmers for use of recovered P fertilisers are needed. Direct economic benefits, increased productivity or profitability seem to be an essential condition for farmers to adopt sustainable practices in the short term (Garbach et al., 2012; Piñeiro et al., 2020).

In regions of intensive livestock agriculture, policies to reduce P losses from manures can indirectly support an increase in P recovery and P recycling. For example, the Dutch government have used an extensive range of policy instruments, in comparison to other countries, to address mismanagement of manures (e.g. the excess application of manure to soils leading to P losses) (Schröder and Neeteson, 2008; Erisman et al., 2011; Backus, 2017). In an overview of Dutch manure management policy instruments from 1984 to 2016, Backus (2017) found restrictions on manure spreading, the requirement to inject manure into the soil, support for flagship farms, and limits on the number of animals were among the most successful and cost-effective measures to reduce nutrient pollution from manures. Between 1980 to 2010, the application of manure P in the Netherlands has been reduced by 50% (i.e. from 160 to 84 kg phosphate as P_2O_5 ha^{-1}) (Backus, 2017). In addition, in 2006, to prevent animal manure from being replaced by mineral P fertiliser, P application limits were extended to both manure and mineral P fertilisers, resulting in a decreased use of mineral P fertilisers and reduced nutrient dispersion into the environment (Malomo et al., 2018). A further outcome of restrictions placed on manure spreading is that many farmers must pay (e.g. crop farmers) for manure disposal (Backus, 2017). In the Netherlands, annual costs for manure disposal in 2007 were estimated at €274 million (CBS, 2016). In 2017, for the average pig farm with no land, costs for manure disposal accounted for ~10% of pig meat production costs (Backus, 2017). The knock-on effect of this has been an increase in the circularity of P flows in agricultural systems, with an increase in farmers' incentives to seek valuable uses of manure, such as processing manure into recovered P fertilisers (Backus, 2017; Malomo et al., 2018). Similar impacts are observed in the US, where since 2006, intensive livestock production has been increasingly regulated. The National Pollutant Discharge Elimination System (NPDES) permitting program regulates the discharge of P to waters, from point sources including concentrated animal feeding operations. Similar to the situation in the Netherlands, polices to reduce P pollution, have led to greater interest in alternative management schemes for further treating or processing manures to make value-added products that can be exported off the farm (e.g., composts or concentrated P products such as struvite; Westerman and Bicudo, 2005).

For many high-income countries, the fertiliser market itself poses a problem, requiring a regulatory framework to provide a level playing field between mineral and recovered P fertilisers (Matsubae and Webeck, 2019). The EU Fertilising Products Regulation aims to level the market for mineral and recovered P fertilisers and help mitigate mineral P fertiliser demand. In June 2019, the European Commission adopted the new Fertilising Products Regulation (EU) 2019/1009, which will apply fully from July 2022 (European Parliment, 2019). This new Fertilising Products Regulation, as a flagship initiative of the first European Circular Economy Package (2015), modernises the conformity assessment and market surveillance in line with the 'new legislative framework' for product legislation. This will mean market access for a wider range of fertilising products, including those manufactured from recovered P materials that were previously excluded (Halleux, 2019), making it easier to sell recovered P fertilisers across the EU, and giving more choice to farmers (European Parliament, 2018). According to the European Commission, the Fertilising Products Regulation will deliver a range of benefits, including the creation of about 120,000 jobs in P recovery, bio-waste recycling, organic fertiliser production and will reduce dependency on PR imports (Halleux, 2019).

Importantly, the Fertilising Products Regulation introduces an initial limit of 60 mg cadmium kg^{-1} P in fertilisers (see Chapter 2). Fertiliser cadmium limits may help boost markets for recovered P fertilisers, as they contain lower/negligible levels of cadmium (1.06-2.30 mg kg^{-1} P_2O_5) in comparison to many mineral P fertilisers (de Boer et al., 2019) and some recovered products have very low levels of impurities and heavy metals. However, bridging the gap between P recovery and actual recycling remains the biggest challenge (Kabbe and Rinck-Pfeiffer, 2019). Part of the issue is ensuring P recovery can produce sufficient volumes of recycled P material; P recovery enterprises are currently much smaller in scale than the mineral P processing industry. Instead of broadening the range of P recovery technologies, investments should be directed towards the development of full-scale demonstrations of the most promising options (Schipper, 2019). Market penetration and replication will only happen with full-scale demonstrations, with the first large-scale operation often requiring some form of government support (Schipper, 2019). Research incentives, among others provided by the EU Horizon 2020 Program, for example, have contributed to funding the development of several pilot technologies that are now ready to be implemented at industrial scales (European Commission, 2019). However, an integrated systems framework should be used to guide decision-making on the most promising options for the local resources and circumstances (Cordell et al., 2011).

The United Nations has adopted global, albeit non-legally binding, normative agreements (e.g. the SDGs and the Paris Agreement) which offer the potential to drive increasing nutrient recovery and recycling (Kanter and Brownlie, 2019). In 2020, the European Union has adopted the European Green Deal and the Farm-to-Fork Strategy for a fair, healthy and environment-friendly food system. These initiatives provide a more favourable framework for P recovery if technologies and products comply with the objectives of high nutrient use efficiency and reducing nutrient losses. In the EU, the post-2020 Common Agricultural Policies include conditionalities, i.e. sustainable practices entitling farmers to premiums. Phosphorus recovery can and should be part of such practices while recovered and PR-derived fertilisers should equally comply with the highest standards to optimise nutrient use efficiency and avoid losses, thus improving water quality outcomes.

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08

Consumption: the missing link towards phosphorus security

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Supporting low levels of animal product (meat, dairy, and eggs) consumption and food waste can significantly reduce the impacts of unsustainable phosphorus use. In addition, consuming products grown with good on-farm nutrient management practices, including phosphorus recycling can further reduce impacts. These changes can contribute to achieving multiple United Nations' Sustainable Development Goals related to improving human and environmental health.

Left: Plant protein burgers cooked with vegan cheese. A reduction in the production of animal products may reduce global agricultural demand and contribute to healthier environments. Image courtesy of likemeat on www.unsplash.com; for further info see www.likemeat.com

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Challenge 8.1: Animal products have high phosphorus footprints

The production of meat, dairy and eggs requires disproportionately high amounts of mineral phosphorus fertilisers. Under 2011 global farming practices, it took 16 times more mineral phosphorus fertiliser to produce 1 g of beef protein than 1 g of legume/pulse protein.

Challenge 8.2: Consumption of animal products is increasing

A 38% rise in the phosphorus footprint of the average diet in the last 50 years is mostly associated with the increased consumption of animal products. A remarkable increase has occurred in China and Brazil; however, their footprints are still below the USA and other industrialised countries (e.g. average per capita protein intake in the EU is about 70% higher than recommended). Economic development correlates with increased consumption of animal products. Some populations still require a more diverse and calorie-rich diet.

Challenge 8.3: Food loss and waste is high across the globe

Globally, 23% of nutrients in fertilisers are used to produce products that are then lost in agricultural and food wastes. The loss at each stage, from farm to fork, differs among regions. Generally, waste is higher on a per-capita basis in industrialised countries, whilst in lower-income countries, losses are driven by insufficient infrastructure.

Challenge 8.4: Changing consumer food habits is difficult

Whilst a shift towards more phosphorus-sustainable diets and waste management practices is required, a complex network of conditions must be met for an individual to change behaviour, which varies by region, country, town, and even family. Raising awareness of negative environmental and/or health impacts (including phosphorus sustainability issues) of certain food choices alone is not enough to change behaviours. People's resources and capacity to change need to be considered as well.

Challenge 8.5: Unsustainable pricing models may slow a transition to sustainable practices

There is a disconnect between what a consumer pays for food and the true 'costs' of food production. The costs involved in mitigating environmental degradation and biodiversity loss from phosphorus losses, and in developing more phosphorus sustainable agriculture systems, are not covered in the price of food products.

Solution 8.1: Reduce consumption of animal products to recommended levels

Wider adoption of healthy diets with low to moderate amounts of meat and dairy (especially low in red meat) could radically reduce demand for mineral phosphorus fertilisers and thus phosphate rock mining. While some demographics could benefit from increased access to animal products, large gains can be made from reducing meat consumption in countries that already consume more than is recommended. The global adoption of a vegetarian diet would cut both fertiliser needs and eutrophication effects by 50%. Although this may be unrealistic, it indicates the major influence of diet change on the global phosphorus cycle.

Solution 8.2: Promote the wide adoption of healthy and regionally appropriate diets

The wide adoption of healthy diets rich in plant-based foods and sustainable aquaculture produce is compatible with sustainable phosphorus management. Sustained communication, along with global and regional structural changes to food systems can help consumers adopt diets that are good for them and the environment.

Solution 8.3: Reduce food loss throughout food production, retail, and consumption sectors

Most food loss in low-income countries occurs before products reach consumers; meanwhile wealthier nations waste more food in retail and at home. Efficient strategies to reduce waste will target the most wasteful, with support underpinned by evidence that quantifies the benefits of change.

Solution 8.4: Make being 'sustainable' easy and rewarding for consumers

It should be easy and affordable for everyone to make healthy diet choices, decrease food waste, and support the safe use of recycled phosphorus from organic wastes (e.g. food waste and excreta) in food production. Incentive structures (including 'health nudges' and 'choice editing') embedded in food systems should be transformed to make phosphorus-sustainable food choices the 'default' option.

Solution 8.5: Develop policies that encourage and support consumers to lead sustainable phosphorus lifestyles

Developing economic and regulatory policies that encourage and support high recycling rates, low animal product consumption and low waste production will be necessary for sustainable change. This may involve setting high goals for organic waste recycling, direct taxes on animal products, or decreasing subsidies that affect the price of meat.

8.1 Introduction

The role that consumers play in the phosphorus (P) cycle is often overlooked. Although most consumers do not physically control how much P is used to fertilise food crops or where their waste goes, they still have great influence over the P cycle through their individual and collective purchasing power, waste management, and through the policies they support. Yet, many consumers feel disconnected from how their food is produced and processed; a trend that is increasing with global urbanisation (Jones et al., 2013).

8.1.1 Individual impacts on phosphorus sustainability

Individual citizens and families affect P sustainability in many ways, however, the largest impact stems from what they eat. Around 85% of all mined P is used in food production (de Boer et al., 2019). Over the last 60 years, 38% of the increased use of mineral P fertilisers can be attributed to global diet changes (Metson et al., 2012). This increase is predominantly related to increased consumption of animal products (meats like beef, poultry and pork, as well as milk and eggs) (Metson et al., 2012, 2016a; Poore and Nemecek, 2018; Li et al., 2019; Oita et al., 2020), especially in wealthier countries where per capita consumption is often higher than is recommended (WHO, 2003). If this trajectory continues, most of the United Nations Sustainable Development Goals will not be met (SDGs) (IPES-Food, 2017; Gordon et al., 2017).

Whilst food consumption is the biggest driver of household P flows, other decisions also have an impact. For example, the maintenance of household sanitation systems (e.g. leaky septic tanks; Withers et al., 2014), the use of lawn and garden fertilisers (Lehman et al., 2011), laundry and dishwashing detergents (van Puijenbroek et al., 2018) and the number of household pets (Chowdhury et al., 2014; van Dijk et al., 2016), can all affect P flows. Also, increasingly affluent and high-tech lifestyles are driving demand for high-grade P in industrial sectors such as steel, iron and battery production (Matsubae et al., 2015) and clothing and construction materials (Hamilton et al., 2018). Indeed, in 2011, 35% of marine and coastal eutrophication and 38% of freshwater eutrophication was associated with the production of non-food products (i.e. clothing, goods for shelter, services and other manufactured products) and these proportions have increased over time (Hamilton et al., 2018).

Phosphorus foot-printing methods have allowed analysis of the P requirements of individuals (Dhar et al., 2021; Metson et al., 2016b; Poore and Nemecek, 2018; Oita et al., 2020), and populations, as well as the P footprint of individual products (Metson et al., 2012). The use of P footprints to assess the sustainability of a given action, behaviour or product should be accompanied by careful analysis of footprint definitions (Čuček et al., 2012). Considerations when interpreting assessments based on P footprints are provided in Focus Box 8.1

Focus Box 8.1 - A closer look at phosphorus footprints

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As consumers, the choices that we make each day leave impressions on our environment. These impressions can be called "footprints" which can be tracked using assessments like life-cycle analysis or material flow analysis. Footprints can be used to compare products for the amounts and sources of P used in their manufacture.

A P footprint is quantified using the inverse of the equation used for P use efficiency (see Chapter 4); it is given in terms of P input or flow per unit of output. Many different footprints can be defined depending on spatial scale, temporal scale, and system boundary.

Foods differ in P footprint, with animal products generally having higher impacts than plant products (Metson et al., 2012). Such footprints can, however, be difficult to calculate. The mined P used in crop production generates fibre and fuel as well as food, and co-products of fuel can transfer P from one production stream to another. For example, dried distillers' grains from ethanol manufacture, rich in P and other nutrients, are consumed by cattle, swine, and poultry. The P in the manure from these animals can support the growth of other crops, including wheat grown for food. The calculation comparing the P footprint of wheat to meat involves allocation assumptions that may need to change when the relative sizes of the different production streams change or as the industries and markets evolve.

The kind of P is as important as the amount. The P input in the footprint could be from mined or recycled sources. Recycled sources could be derived from animal manures, food waste, sewage, or other sources. Another useful but different definition of the footprint might involve the amount of P lost to drainage water per unit of agricultural production.

As a sustainability metric, P footprints should be considered in balance with others. For example, the "field-print" defined by the Field to Market Alliance for Sustainable Agriculture includes biodiversity, energy use, greenhouse gas emissions, irrigation water use, land use, soil carbon, soil conservation and water quality. Phosphorus footprints need to be considered in the context of these other metrics, selected for their priority to the stakeholders of the food value chain (Field to Market, 2018). They should also be considered in the context of the footprint of other nutrients including nitrogen and potassium.

8.1.2 The capacity of consumers to support phosphorus sustainability varies

Globally, the human population is increasing. However, stabilising this increase may not significantly improve P sustainability as high per capita consumption and pollution needs to be addressed (Vörösmarty, 2000; Bapna, 2011). The exponential growth of 'middle class' populations around the world complicates matters because increased wealth has historically meant greater resource demand, and this cannot continue indefinitely (Bapna, 2011).

To better understand sustainable phosphorus behaviours, we must better understand what motivates food consumption and waste management behaviour. For instance, social norms play a large role in why increased income and urbanisation in China has translated to a large increase in the consumption of animal products (Zhai et al., 2014); the aspiration towards a Western diet (and arguably Western waste management systems) is a powerful driver even if those Western systems are not sustainable. Individual food consumption behaviours are also sometimes determined by religion (Pechilis and Raj, 2012). For example, most followers of the Hindu faith are lacto-vegetarians (e.g. exclude meat, fish, poultry and eggs) and followers of the Buddhist faith are strict vegetarians, while the Christian faith does not have any rules regarding food choice (Kittler et al., 2016). If wealthy, and/or socially powerful, consumers and organisations set dietary norms to be less animal product intensive, then it may be possible to decouple increasing wealth with resource- and waste-intensive lifestyles. Re-imagining what it is to live a good life within planetary boundaries requires rethinking our social boundaries, including equity and justice (Brand et al., 2021).

Importantly, not all consumers have the same financial, infrastructural, and social resources to support sustainable P management. In some contexts, reducing animal product consumption is neither desirable nor possible due to serious health concerns (e.g. childhood stunting) (Kaimila et al 2019) and/or a lack of affordable, accessible, healthy alternatives (Widener 2018). Similar concerns arise when considering access to sanitation and waste management options (e.g. Öberg et al., 2020), which can affect P recycling. Consumers (primarily, but not exclusively, in the Global North) who do have the capacity for sustainable P lifestyles can directly reduce P demand and pollution with their choices, and indirectly support P security by affecting global food supply chains and social norms.

In this chapter, we argue that although different strategies will be required for different regions, the goal is the same: to support both environmental quality and human health in the long term through better consumption practices. This chapter highlights that the public, as food consumers, waste producers, and decision-makers, play a critical role in the sustainability of the P cycle. However, sustainable products should be readily available and affordable for consumers to choose from, which in most cases will require greater collaborative efforts among policymakers, institutions (e.g. schools, hospitals), and food processing, distribution, and retail services (e.g. restaurants). In the following sections, we summarise the key challenges to increasing phosphorus sustainable consumption behaviours and suggest potential solutions to overcome them.

8.2 The Challenges

Challenge 8.1: Animal products have high phosphorus footprints

The production of meat, dairy and eggs requires disproportionately high amounts of mineral phosphorus fertilisers. Under 2011 global farming practices, it took 16 times more mineral phosphorus fertiliser to produce 1 g of beef protein than 1 g of legume/pulse protein.

Each crop and animal has particular nutrient needs. Thus, the foods that compose our diets affect how much P is used in food production (Figure 8.1) (Metson et al., 2012). In general, animal proteins (especially beef) require larger inputs of P to produce than legume/pulse protein (Dhar et al., 2021, Metson et al., 2012, 2016a; Poore and Nemecek, 2018; Li et al., 2019; Oita et al., 2020). This is because animals require not only a certain amount of P, but large amounts of feed crops to meet carbohydrate and protein needs, and these feed crops also require P to grow. Because there are more steps in animal production than plant production, animals are associated with larger P losses to waterways (see Chapter 5). Under 2011 global farming practices, it took 16 times more mineral P fertiliser to produce a gram of beef protein than a gram of legume/pulse protein (Metson et al., 2012). This is a conservative estimate because it assumes that grasslands and pastures are not fertilised (other than with recycled manure) which is not currently the case in many areas (e.g. North-Western Europe, Australia) and is unlikely in the future (Sattari et al., 2012). However, this assumption cannot be applied in some countries like Malawi, where 80% of livestock feed comes in the form of unfertilised pasture and 50% of the excreta remains on the land (Mnthambala, 2021). Pork, chicken, milk and egg production all require less P per unit of protein than beef. However, producing one unit of animal protein still takes up to ten times the resources (not just P) of producing one unit of vegetarian protein (White and Cordell, 2015). The dependence of animal production on mineral P fertilisers can be reduced by optimising the recycling of manure and other organic waste in the production of animal feed. Still, some losses are unavoidable, and there are multiple challenges to safe, economical and agronomically appropriate recycling that should be addressed (discussed in Chapters 6 and 7). Farmed blue food (fish and other aquatic foods from freshwater and marine environments) can also cause leakage of P, but most systems emit slightly less than poultry (in terms of kg of edible yield) (Gephart et al. 2021). For fed aquaculture, 94% of emissions stem from on-farm production (Gephart et al. 2021). In some circumstances, non-fed aquaculture, such as mussels and seaweeds, are extractive systems that remove P from the water body and can therefore be considered part of the solution to eutrophication. Blue food from capture fisheries causes no emissions of phosphorus.

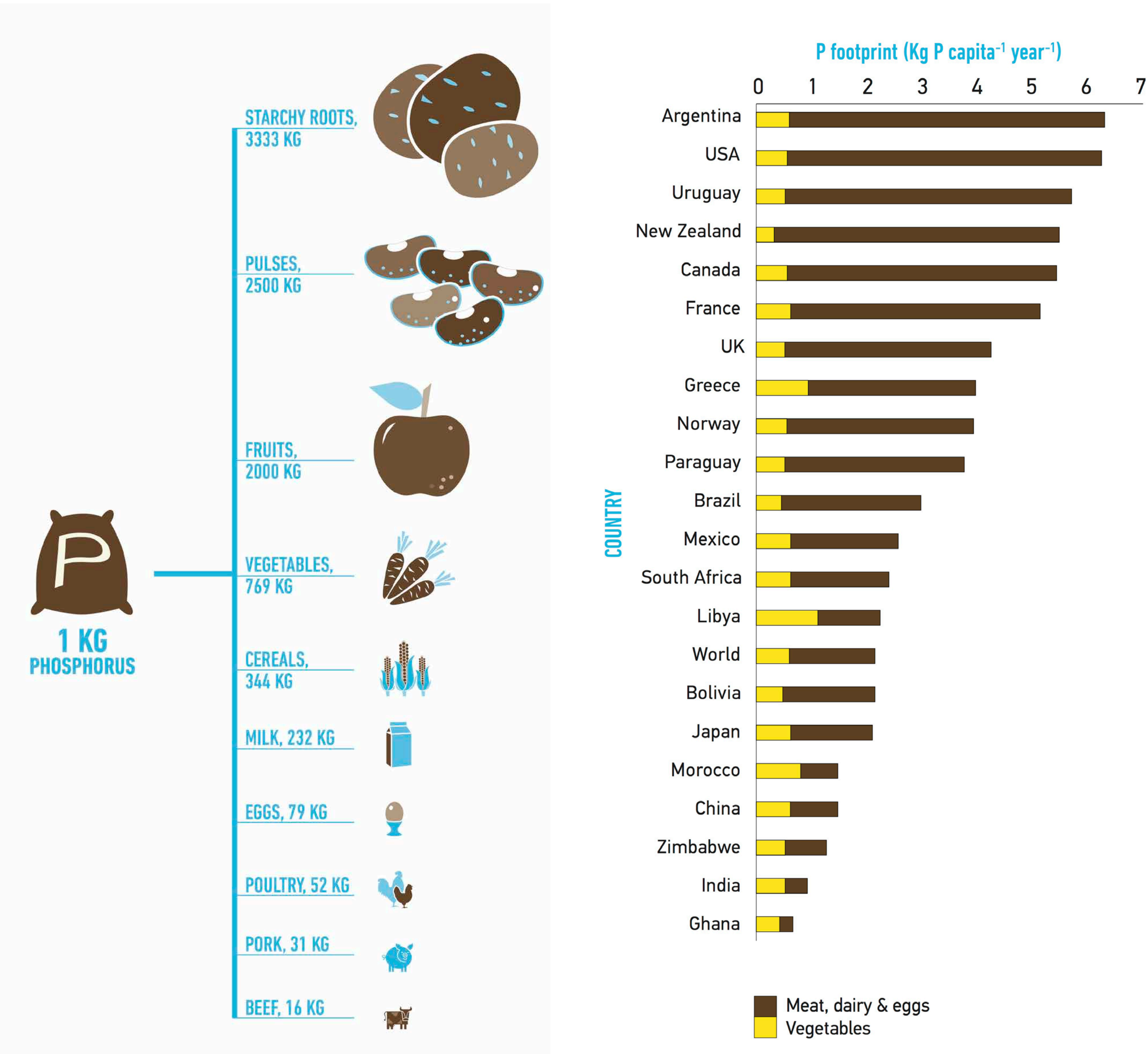

Figure 8.1 Dietary phosphorus (P) footprint associated with different food groups showing that animal products require more mineral fertiliser than plant crops (left side of figure), and the wide range of P footprint values across countries which are driven by meat consumption (right side of figure). The P footprint values for a country are expressed as the average amount of mineral P fertiliser required to produce food for one person for one year in that country given 'current' global agricultural practices. Reproduced with permission from HEADWAY (2013) with data based on Metson et al. (2012).

Challenge 8.2: Consumption of animal products is increasing

The average diet has seen a 38% rise in phosphorus footprint over the last 50 years; this can mostly be attributed to the increased consumption of animal products. A remarkable increase has occurred in China and Brazil; however, their footprints are still below the USA and other industrialised countries (e.g. average per capita protein intake in the EU is about 70% higher than recommended). Economic development correlates with increased consumption of animal products. Some populations still require a more diverse and calorie-rich diet.

Despite a significant increase in the consumption of animal products in countries such as Brazil and China over the last 20 years, levels of consumption are still well below those of North American and most other industrialised countries (Westhoek et al., 2015). The current average per capita protein intake in the European Union (EU) is about 70% higher than necessary according to the World Health Organization (WHO) recommendations (WHO, 2003).

It is important to note that there are still undernourished populations that require more calories and a more diverse diet (Alexandratos and Bruinsma, 2012). In many countries across the Global South, a lack of animal protein is responsible for stunting in children under five and their diets must be artificially supplemented (often with milk powder) (Kaimila et al 2019). Animal products can provide high-value protein and essential micronutrients (i.e. iron and zinc, and vitamin A). However, high consumption of animal products, in particular red meat, in some countries and social classes has led to significant health issues (Alexandratos and Bruinsma, 2012) and global environmental damage (Stenfield et al., 2006; Machovina et al., 2015).

Economic development correlates with increased consumption of animal products. As a generalisation, as incomes increase, people tend to eat more meat (Stamoulis et al., 2004; Keats and Wiggins, 2014). For example, as China and India, which together account for 37% of the global population, have gained wealth and become increasingly urbanised, there has been a shift from a cereal-based diet to more animal products (meat, eggs, dairy, fish) as well as fresh fruits and vegetables (Gandhi and Zhou, 2014). Per capita income increased by over 1,000% in China from the early 1980s to 2010, accompanied by rural (300%) and urban (166%) increases in meat consumption, though meat consumption was already higher in urban areas (Gandhi and Zhou, 2014). Today China is the largest consumer of meat in the world (Godfray et al., 2018).

Increases in mineral P fertiliser consumption were significantly correlated with increases in meat consumption in China between 1950 and 2010 (Bai et al., 2016). China's P footprint increased by 400% between 1970 and 2010 (Metson et al., 2012). If the highest population projections become a reality, and global diets continue to shift towards more meat and more calories, by 2050 demand for P fertilisers could increase by 141% from

2007 levels (Metson et al., 2012). Under current production systems, animal products are associated with higher eutrophication potential per serving (Poore and Nemecek, 2018; Willett et al., 2019). Without changes to production practices, increased consumption will lead to more P pollution.

High consumption of dietary P is linked to eating more processed foods that use P as an additive (León et al., 2013), which can cause serious health problems for people with kidney disease (González-Parra et al., 2012). Prevalence of chronic kidney disease (defined as a reduced glomerular filtration rate, increased urinary albumin excretion, or both) is estimated to be 8–16% worldwide (Jha et al., 2013). Kidney disease is an increasing public health issue; the prevalence of end-stage renal disease in the USA population has been predicted to increase by 48% during the next decade and will pose a significant health cost burden (Nickolas et al., 2004; Jha et al., 2013). The increasing use of P additives in food could be problematic for people who do not know they have kidney problems and is complicated by the fact that food is rarely labelled for total P content, making it hard to avoid (Uribarri and Calvo, 2017). Awareness of the disorder remains low in many communities and physicians (Jha et al., 2013).

Challenge 8.3: Food loss and waste is high across the globe

Globally, 23% of nutrients in fertilisers are used to produce products that are then lost in agricultural and food wastes. The loss at each stage, from farm to fork, differs among regions. Generally, waste is higher on a per-capita basis in industrialised countries, whilst in lower-income countries, losses are driven by insufficient infrastructure.

Globally, 23% of the nutrients in fertilisers (P, nitrogen, and potassium) are used in products that are lost in food loss and waste (Kummu et al., 2012). While large amounts of food waste in Asia can be attributed to the large population, food waste is much higher on a per-capita basis in industrialised countries than in low-income countries (Kummu et al., 2012). Consumers in Europe and North America waste 95-115 kg $year^{-1}$ of food, in contrast to only 6-11 kg $year^{-1}$ in Sub Saharan Africa and South/ South-East Asia (Gustavsson et al., 2011).

The amount of food loss or waste at each stage, from farm to fork, also differs across regions. In general, lower-income countries have more food loss before products reach consumers because of food storage issues, while wealthier nations tend to waste more food in retail and home settings (Parfitt et al., 2010). Therefore, interventions to reduce food loss and waste across regions may differ significantly.

A study in the US suggested that diets that are rich in fresh fruits and vegetables are linked to higher amounts of waste because these foods can easily perish

(Conrad et al., 2018). However, because fruits and vegetables have a lower fertiliser footprint than many other foods, this diet can still contribute less to the onset of eutrophication (Poore and Nemecek, 2018). Cultural aspects of diet, in conjunction with religious rituals, are also known to contribute to the problem of food loss. During Ramadan in some Arabic countries, almost 30%-50% of the food prepared is wasted because of excessive meal preparation (Abiad and Meho, 2018). Similarly in India, nearly 8 Mt waste year^{-1} is produced from temple, mosque, and church offerings (ASK-EHS, 2019) which usually include milk, fruits and sweets along with flowers and tree leaves. These sacred offerings are frequently thrown into rivers, ponds and lakes where they can cause significant harm to water ecosystems (ASK-EHS, 2019). To achieve the maximum environmental and resource benefits, the potential of food waste and diet change should be considered together in the context of complex cultural norms.

Challenge 8.4: Changing consumer food habits is difficult

Whilst a shift towards more phosphorus-sustainable diets and waste management practices is required, a complex network of conditions must be met for an individual to change behaviour, which varies by region, country, town, and even family. Raising awareness of negative environmental and/or health impacts (including phosphorus sustainability issues) of certain food choices alone is not enough to change behaviours. People's resources and capacity to change need to be considered as well.

People do not make decisions based on a single criterion, which complicates finding strategies that address the needs of all consumers (Vermeir and Verbeke, 2006). A complex network of conditions must be met for an individual to change behaviour; spanning from individual- and household-level factors to more slow-changing contextual factors, which all shape our decisions (Schill et al. 2019). However, pro-environmental behaviours are often significantly influenced by social norms (Nyborg et al. 2016; Farrow et al. 2017) as well as habits, rather than reasoning (Klöckner and Matthies, 2004; Klöckner, 2013).

This is particularly the case for decisions about what to buy, cook and eat. Such weekly (or even daily) decisions are influenced by habit strength or simply the wish to have a convenient meal that everyone around the table likes (Ouellette and Wood 1998; Klöckner and Matthies, 2004; Nilsen et al., 2012; Nyborg et al. 2016). Therefore, to achieve more P sustainable behaviours, we must address how social norms and habits are created, reinforced and continued (Klöckner and Matthies, 2004; Nyborg et al. 2016).

Education about environmental problems may increase an individual's level of concern, but such concern is generally not sufficient to change behaviour (Kollmuss and Agyeman, 2002; Bamberg, 2003; Barr, 2004). Bamberg (2003) showed that environmental concern accounted for less than 10% of the variance in environmental behaviour in the combined meta-analyses of Hines et al. (1987) (128 studies) and Eckes (1994) (17 studies). However, conscious (rather than habitual) decisions to reduce consumption of animal products based on other motivations, such as health or animal welfare (Fox and Ward, 2008; de Boer et al., 2017), are in most cases also directly beneficial for P management. Attitudes and behaviours related to waste management, and acceptance of recycled organic residues as a fertiliser can also be difficult to change (Chapters 4 and 6).

Challenge 8.5: Unsustainable pricing models may slow a transition to sustainable practices

There is a disconnect between what a consumer pays for food and the true 'costs' of food production. The costs involved in mitigating environmental degradation and biodiversity loss from phosphorus losses, and in developing more phosphorus sustainable agriculture systems, are not covered in the price of food products.

In the UK, the 'hidden costs' of food production (which can include environmental degradation, biodiversity loss, diet-related disease, farm support payments, regulation and research) would almost double the price of food under current agricultural management and food purchasing habits (Fitzpatrick et al., 2019). Over a third of unaccounted costs (£45 billion out of the £120 billion estimated for 2015) are related to natural capital degradation and the loss of biodiversity and ecosystem services. This includes water pollution and wasted food, which is in part related to P management and sustainability. In many cases, P pollution is not sufficiently managed, and society pays the price with declining ecosystem services (e.g. recreational services, drinking water, ecosystem quality (Pretty et al., 2003; Dodds et al., 2009) (see Chapter 5).

Food prices rarely, if ever, cover costs of practices that would increase resilience to fluctuations in the availability of mineral P (phosphate rock and/or fertilisers).

For example, in 2008 the price of food skyrocketed (e.g. rice prices doubled within five months, up to US$757 t^{-1}) due to multiple stressors including energy prices, drought, and market speculation (Baffes and Haniotis, 2010) (see Chapters 2 and 3). This was accompanied by an 800% increase in the price of P fertilisers (Cordell and White, 2014) (see Chapters 2 and 3). This spike disproportionately affected poorer farmers (whose farm budget is often mostly spent on fertilisers) and poorer consumers for whom food is a higher part of overall household budgets. After 2008, however, there was no large shift towards investment in alternative or more diversified sources of P, and so communities remain vulnerable to such shocks (Cordell et al., 2015).

Investment in P recycling could help reduce food security risks associated with imported mineral P fertilisers (see Chapters 2, 6 and 7). Few countries, regions, and cities have set goals to minimise P waste and increase recycling; where goals exist, successful large-scale implementation remains limited (Metson and Bennett, 2015; Kabbe, 2019). A lack of waste collection and processing technologies and infrastructure are major barriers to recycling and recovery (see Chapters 6 and 7). These issues, however, are often underpinned by a lack of public support, laws and regulations, and unfavourable cost-benefit analyses (Drechsel et al., 2010; Withers et al., 2015; Seufert et al., 2017; Metson et al., 2018; Öberg and Mason-Renton, 2018).

8.3 Solutions

Solution 8.1: Reduce consumption of animal products to recommended levels

Wider adoption of healthy diets with low to moderate amounts of meat and dairy (especially low in red meat) could radically reduce demand for mineral phosphorus fertilisers and thus phosphate rock mining. While some demographics could benefit from increased access to animal products, large gains can be made from reducing meat consumption in countries that already consume more than is recommended. The global adoption of a vegetarian diet would cut both fertiliser needs and eutrophication effects by 50%. Although this may be unrealistic, it indicates the major influence of diet change on the global phosphorus cycle.

Lowering global consumption of meat, dairy and eggs could radically reduce the use of mineral P fertilisers. Producing a vegetarian's diet requires 1.0 kg P $year^{-1}$ less than for a meat-eater (Elser and Bennett, 2011). If all humans adopted a strictly vegetarian diet, it would decrease mineral P fertiliser needs by at least 50% (Metson et al., 2012), which could reduce eutrophication by 49% (37-56%, based on the current ‘best’ or ‘worst’ practices for vegetable protein production (Poore and Nemecek, 2018).

Figure 8.2 A meat market in China. Meat consumtion has increased significantly in China and Brazil in the last 20 years; however, their phosphorus footprints and average per capita meat conusmption are still below the USA, the EU and many industrialised countries.

Diets with moderate dairy and meat consumption can also improve health and average life spans while reducing global warming impacts (Tilman and Clark, 2014). That said, it is not realistic, or necessarily desirable that the entire human population would adopt a vegetarian or plant-based diet. Average global meat consumption is estimated at ~43 kg capita^{-1} year^{-1}, with consumption in high-income countries roughly double this at ~85 kg capita^{-1} year^{-1} (data for 2013; FAOSTAT, 2018). Defining how much meat or dairy should be considered 'low' or 'moderate' consumption depends on individual circumstances (e.g. the size of person, their diet and activity levels) and regional social and environmental context. In a study exploring how to sustainably feed Nordic populations, Karlsson et al. (2017) suggested a sustainable diet should contain between 80 and 150 g of meat capita^{-1} week^{-1} (~30 kg meat capita^{-1} year^{-1}). The EAT-Lancet Commission recommends 0-196 g of red meat capita^{-1} week^{-1} (Willett et al. 2019). Resare Sahlin et al. (2020) argue that the research community needs to provide a more informed explanation to consumers of what is 'less' and what is 'better' when providing guidance on meat consumption.

It is also important to note that although beef generally requires much more mineral P fertiliser than other animal products (Metson et al. 2012), this does not mean that it is the only product that needs to be consumed in moderate amounts. In fact, under a scenario where only pastures and food waste are used to feed animals in Nordic countries, overall meat consumption would need to decrease but the proportion of beef consumption could increase slightly (Karlsson et al., 2017). Nevertheless, low animal product diets are an essential part of P sustainability, whilst decreasing food waste and supporting safe recycling and sustainable farming can provide opportunities for a diversified food production system.

Solution 8.2: Promote the wide adoption of healthy and regionally appropriate diets

The wide adoption of healthy diets rich in plant-based foods and sustainable aquaculture produce is compatible with sustainable phosphorus management. Sustained communication, along with global and regional structural changes to food systems can help consumers adopt diets that are both good for them and the environment.

Low to moderate consumption of meat, dairy, and egg consumption is in line with guidelines for healthy diets (Westhoek et al., 2015; Willett et al., 2019). High levels of processed meat consumption are associated with higher rates of colorectal cancer (Godfray et al., 2018), while low-animal, vegetarian and pescatarian (blue-food) diets have been associated with a lower incidence of type 2 diabetes, cancer, and death related to cardiovascular issues (Tilman and Clark, 2014).

Accompanied by increased yields from judicious P application and recycling, low animal product consumption and low food waste globally could reduce environmental degradation and feed more people adequately and sustainably. Converting lands that currently produce livestock feed and biofuels to crops for human consumption could produce food for an additional 4 billion people (Cassidy et al., 2013). In low-income countries, increasing recycling could help close yield gaps and contribute to food security (Dumas et al., 2011; Akram et al., 2018).

Using land and feed resources that do not compete with calories produced for direct human consumption could supply 15 to 46% of protein requirements per person per year, globally (van Zanten et al., 2018). To contribute to food system sustainability, however, countries that consume large amounts of animal products would have to decrease their current levels of consumption so that countries, where consumption is low (e.g. in parts of Asia and Africa), could moderately increase their consumption (van Zanten et al., 2018). Among populations that consume little protein such as in Malawi, crops like maize are culturally important but are not adapted to the local environment. Maize requires significant P inputs (e.g. 21 kg P ha^{-1}) and offers little nutrition. Encouraging the cultivation and consumption of more nutritious and well-adapted plants like cassava or sorghum (which require no additional P) could have an important impact on health and P use (Government of Malawi, 2012).

In summary, a globally sustainable and healthy diet would consist of vegetables, fruits, whole grains and vegetable proteins, with small amounts of animal products, blue food, and processed foods (Willett et al., 2019). To be environmentally sustainable, these foods should be produced in a way that minimises food waste and maximises nutrient recycling, including P (Willett et al., 2019). The specific composition of a healthy diet, the production practices that enable it, and who bears responsibility for change needs to be regionally specific. However, in all cases food production should embrace principles of equity and acknowledge differences in power and capacity to influence change (Hirvonen et al., 2020; Moberg et al., 2020; Mui et al., 2021).

Solution 8.3: Reduce food loss throughout food production, retail, and consumption sectors

Most food loss in low-income countries occurs before products reach consumers; meanwhile wealthier nations waste more food in retail and at home. Efficient strategies to reduce waste will target the most wasteful, with support underpinned by evidence that quantifies the benefits of change.

Food waste could be halved if every country had the lowest level of loss and waste currently achievable at each step of food production (Kummu et al., 2012). Eliminating consumer food waste alone for wheat, rice, vegetables and meat in the USA, India, and China could free up enough calories to feed over 413 million people year^{-1} (West et al., 2014), and could simultaneously reduce P application to fields and losses to waterways. Avoidable food waste in the USA in 2009 had a retail value of almost US$198 billion; 63% of that value was due to losses at the consumer level (Venkat, 2012). On the other hand, with limited energy for refrigeration, poor transportation networks, and prohibitive trade barriers, much of the seasonal crops in less economically developed tropical countries goes to waste: innovation is required to help producers reach markets before the next glut of mangos or avocados is left to rot, despite having eager buyers in the North (Affognon, 2015).

Eliminating consumer food waste would deliver economic benefits. For example, if EU households reduced food waste, it could yield annual household savings of €123 per capita (40% reduction by 2020), 7% of the average annual EU household budget spent on food (Rutten et al., 2013). Across the EU, this amounts to an annual saving of €75.5 billion. Reducing food waste by 40% in households and 60% in retail in the EU would free up 28,940 km^2 of agricultural land, equivalent to the land area of Belgium (Rutten et al., 2013). These savings could be used to purchase more expensive foods produced with good P management practices (including safe and effective recycling) from farms that may not be currently economical.

However, not all losses are easily avoidable, for example, the inedible parts of crops and animals, although recycling options exist for both (see Chapters 6 and 7). In addition to unavoidable losses before food is consumed, we should also consider post-consumption losses of P from animal and human excreta. All organic waste sources can theoretically be recycled. Food waste and human excreta will continue to accumulate in cities as populations urbanise and grow. Supporting behaviours and technologies that allow for P recycling will thus be essential for sustainable agriculture and limiting eutrophication (Willett et al., 2019; van Puijenbroek et al., 2019). Recycling P in the right amounts to achieve maximum yields is potentially one of the greatest opportunities to decrease mineral P fertiliser application rates (Springmann et al., 2018), but this requires large changes to current food production systems, as well as public support (see Chapters 6 and 7).

Solution 8.4: Make being 'sustainable' easy and rewarding for consumers

It should be easy and affordable for everyone to make healthy diet choices, decrease food waste, and support the safe use of recycled phosphorus from organic wastes (e.g. food waste and excreta) in food production. Incentive structures (including 'health nudges' and 'choice editing') embedded in food systems should be transformed to make phosphorus-sustainable food choices the 'default' option.

To reduce the P requirements of consumers', it is necessary to identify which behaviours contribute to their P footprint, as well as the factors that shape those decisions. Then, barriers to behaviours that promote sustainable P management can be removed and opportunities harnessed, while healthy dietary requirements can still be met, and safe waste handling achieved.

Common to many models of environmental behaviour is the understanding that interventions must tackle the conscious and unconscious parts of decision making (Baranowski et al., 2003; Klöckner, 2013; Marteau, 2017; Godfray et al., 2018). Education campaigns and labelling (Leach et al., 2016) can be part of these interventions, but are not be sufficient on their own (Gordon et al., 2017; Poore and Nemecek, 2018, Röös et al., 2021). Education about the detrimental effects of high meat consumption on the environment (or other issues such as animal welfare) may increase the intention to reduce these behaviours but rarely results in actual behavioural change (Bianchi et al., 2018a). Interestingly, eco-labels are shown to work better in countries where there is more state control (Sønderskov and Daugbjerg, 2011). To make education more effective, tracking behaviour over time can help (Bianchi et al., 2018a), but sustainable products also need to be available (Bianchi et al., 2018b) and affordable (Widener 2018). Similarly, an increase in the supply of sustainable and healthy alternatives must be supported with education to underpin demand (Allcott et al., 2018). Regarding food waste, some hopeful results stemming from a longitudinal field experiment in Sweden, found a significant increase in recycled food waste following a household-targeting information campaign about food waste recycling (based on insights from nudging and community-based social marketing) and could inform similar pro-environmental behaviour interventions elsewhere (Linder et al. 2018).

More important, perhaps, is the perception that sustainable products are available, affordable, and part of the social norm (Vermeir and Verbeke, 2006; Nyborg et al. 2016). Explanations for differences in what people buy to eat across countries and regions are typically a combination of available incomes, food prices but also nutritional content and norms (Dubois et al. 2014; Nyborg et al. 2016). For some people, it is also important to feel their actions are desirable (a reward) while also making a difference (reflection) (de Boer et al., 2018; Vermeir and Verbeke, 2006). Similarly, for farmers to participate in sustainable P management schemes, they must feel that their actions are meaningful, and perhaps most importantly, that such actions match their values and those of their community (Chapman et al., 2019).

Donner (2017) suggests that sustained communication across a variety of platforms and audience-specific frames may be the best way for governments, businesses, and other organisations (e.g. schools, hospitals) to increase the relevance of environmental science to policy and the public. Successful examples of dietary change include awareness raising and education campaigns in South Korea, which focused on increasing the consumption of low-fat high-vegetable meals (Keats and Wiggins, 2014). This approach could be successful in other countries if policies supported pricing mechanisms that benefited plant-based agriculture, especially legumes, fruits and vegetables, rather than livestock and grains, as has been the case historically (Clonan et al., 2015). Interventions can also involve making more P-sustainable 'default options'. Decreasing the size of meat portions and increasing vegetable sizes in restaurants and cafeterias have successfully increased P sustainability without affecting customer satisfaction (Reinders et al., 2017; Wynes et al., 2018; Bianchi et al., 2018a). Systematic changes that make sustainable dietary choices easy, desirable, and affordable are more likely to produce lasting change.

Solution 8.5: Develop policies that encourage and support consumers to lead sustainable phosphorus lifestyles

Developing economic and regulatory policies that encourage and support high recycling rates, low animal product consumption and low waste production will be necessary for sustainable change. This may involve setting high goals for organic waste recycling, direct taxes on animal products, or decreasing subsidies that affect the price of meat.

To see the magnitude of change needed, the incentive structures embedded in food systems must be transformed (Oliver et al., 2018) (see Chapter 3). In industrialised food systems, power has become increasingly concentrated in a small number of large companies (Folke et al., 2019; Gordon et al., 2017; Godfray et al., 2018), meaning that directed interventions on a few key actors could have large and lasting effects throughout the system. Local cultural and resource contexts also need to be considered (Bere and Brug, 2009). Interventions that do not centre on individual choices to reduce meat consumption or waste, but rather the system that shapes those decisions, appear to be more successful (de Boer and Aiking, 2018). This should include policies that affect powerful actors, which can have cascading effects on large numbers of consumers and producers (Clapp 2018).

Governments could set high goals for organic waste recycling, directly tax animal products, implement a carbon tax that indirectly affects animal products or decrease subsidies that affect the price of intensively produced animal protein. All of these option could change incentive structures to decrease meat consumption and increase the recycling of P and other nutrients. Policies that target food system intermediaries, such as processors, distributors and retailers, will be essential since they have more direct contact with both farmers and consumers (Canning et al., 2016). Attention must also be given to the development and enforcement of policies that affect food producers in global food supply chains, most notably those that are using recycled P sources. For example, Kenyan growers who are allowed to use excreta in agriculture (within national frameworks) are prohibited from exporting their products to the EU (Moya et al, 2019).

Municipalities, certification organisations, and businesses can adopt policies and infrastructure that support systemic changes to food and waste sectors. For example, in Canada and the EU, cities that invested in centralised infrastructure for separated organic waste collection and economic disincentives for landfilling have diverted high amounts of organic waste from landfills, thereby recycling P to agricultural production (Treadwell et al. 2018). On the other hand, regulations can inhibit the recycling of P, for example banning human excreta from organic production (Seufert et al., 2017) or banning the reuse of animal bones at the EU level. Government agencies can also incentivise individual behaviours. In the USA, for example, providing participants in food assistance programmes with a voucher for fresh fruit and vegetables increased purchases, and reduced the gap between actual and recommended consumption of fruit and vegetables by 20%, compared with restricting purchases of unhealthy products or no intervention (Olsho et al., 2016). These interventions do not take away options but instead create a consumer environment where it is easy to make a sustainable choice, whilst benefiting business.

Fortunately, the changes in the food system required for P sustainability also align with many of the changes that are required to meet other societal goals. As such, interventions are likely to have multiple benefits, including both human (see Challenge 8.2 and Solution 8.2) and environmental health. For instance, producing animal protein requires more resources and causes more environmental harm than plant-based protein (Clune et al., 2017; Hilborn et al., 2017; Poore and Nemecek, 2018), not just associated with phosphorus. A 100% plant-based diet could reduce land use by 76%, would halve greenhouse gas emissions, acidification, and eutrophication, and would reduce freshwater withdrawals associated with the food system by 19% (Poore and Nemecek, 2018). Even a 50% cut in livestock production could make a huge impact. In the EU, for example, it would mean a 40% reduction in greenhouse gas emissions and reactive nitrogen use in the agricultural sector and a 75% reduction in soybean imports (Westhoek et al., 2014).

The challenges identified above must be tackled across different scales because patterns of food consumption and waste production stem from decisions and actions of policymakers, institutions (e.g. schools, hospitals), businesses (e.g. food processors, grocery stores and restaurants), households and individuals. They occur in the context of a diversity of local to global policies, infrastructure, and cultures. Different stakeholders need to participate collectively in making changes for better P management (Table 8.1).

Although not all specific local interventions will be win-wins without careful planning, the three major changes proposed here are in line with the changes required globally for a better food system (Figure 8.2).

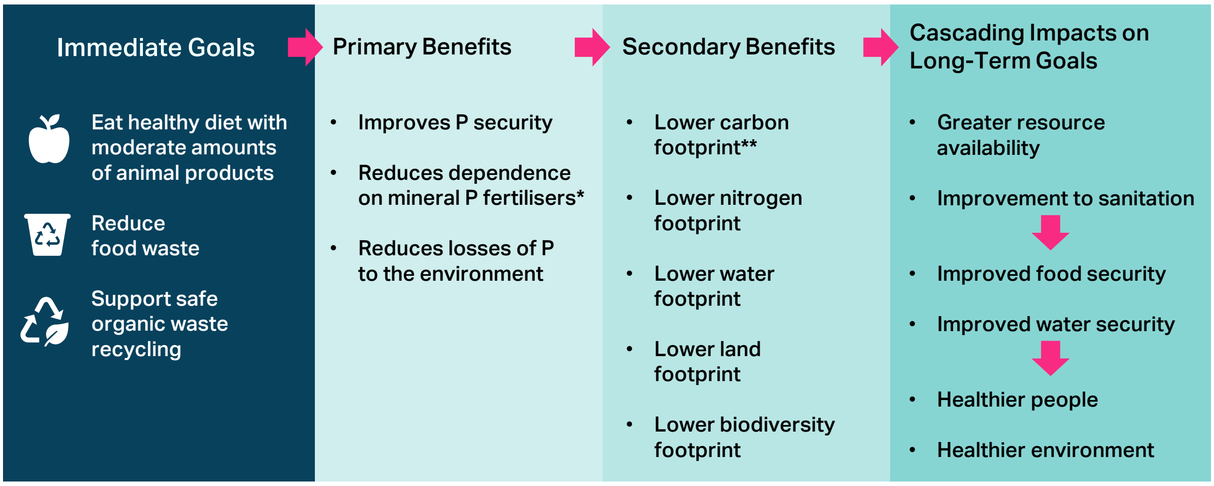

Figure 8.3 Benefits of the goals and interventions recommended by this Chapter (*we highlight fertiliser use is still encouraged where appropriate and **lowering the carbon footprint of dietary consumption can contribute to a reduction in climate change impacts).

Table 8.1 Examples of actions that different actors can take to enhance phosphorus (P) sustainability/security, in support of the three goals (left side of the table) related to food consumption.

		Actors			
		National, regional & municipal policymakers/regulators	Certification bodies and stand-ards organisation	Food processors, distributors, and retailers (including restaurants, cafeterias, and catering)	Citizens/consumers
Goals	**Eat healthy diets with moderate amounts of animal products**	Tax on animal products. Finance education and research on communicating healthy low impact diets. Eliminate subsidies for food produced in non-environmental ways. Create coupons for food items that have low environmental impacts.	Set food guides that reflect the latest research.	Increase the production and availability of vegetarian and pescatarian options. Offer default vegetarian meals. Decrease animal product portion size.	Buy fewer (and better) animal products. Eat more healthy plant-based whole foods. Create advocacy groups and vote for people and policies supporting access to healthy food for all.
	Reduce food waste	Set waste reduction goals. Finance education campaigns on waste reduction. Support local food systems that bring consumers and producers closer together.	Enforce penalties on large waste producers. Implement tools for tracking products and their environmental impacts.	Invest in proper (cold) storage facilities. Buy in the correct quantity. Allow the sale and use of 'imperfect' but safe agricultural produce.	Do not buy more than you can eat or freeze. Buy from shops and restaurants that have a low/zero waste policy.
	Support safe organic waste recycling	Set recycling targets and laws. Invest in source separation and waste collection infrastructure. Finance research in recycled products development. Finance education campaigns on the importance of recycling.	Create labels that indicate good nutrient management (including recycling). Require the use of recycled P in ecologically certified products. Develop and enforce certification standards for recycled P fertiliser.	Eliminate packaging that makes recycling difficult. Set purchasing policies that require good nutrient management (including recycling). Enforce separate organic waste collection in buildings.	Create advocacy groups and vote for people and policies supporting P recycling. Live in, or advocate for, buildings and cities that invest in source separation of waste (e.g. excreta, food, and garden waste). Buy products that are certified for good nutrient management.

This means that communities can, and should, harness the fact that goals other than P may be stronger motivations for change. For example, reducing meat consumption in Sweden by 50% and replacing it with Swedish-produced legumes would allow citizens to meet healthy diet recommendations (SDG 2), decrease greenhouse gas emissions (SDG 13), and free up 21,500 ha of land to meet other national goals including biofuel production (SDG 14) or nature conservation (SDG 15) (Röös et al., 2018). With judicious planning, better animal welfare (which currently does motivate plant-based and low-meat diets) could also support P sustainability. Less demand for meat, eggs, and dairy could allow a transition away from highly specialised, concentrated animal production systems towards integrated animal and crop production systems (Robbins et al., 2016). Shorter transport distances between areas where animals are born, raised, and slaughtered are beneficial for animal welfare (Schwartzkopf-Genswein et al., 2012), as animals can spend more time outside. This could improve P management, as manure transport is often a barrier to recycling (Westerman and Bicudo, 2005; Keplinger and Hauck, 2006; Nicholson et al., 2012).

Improving P sustainability by reducing animal product consumption, reducing food and agricultural waste, and increasing organic-waste recycling will help deliver multiple United Nations Sustainable Development Goals (SDGs). For instance, achieving universal sanitation (SDG 6) with nutrient recovery and energy recovery technology could meet approximately 9% of P fertiliser demands (SDG 2) and 1% of household energy needs (SDG 13), with larger gains in areas with current low access to sanitation infrastructure (Trimmer et al., 2017). Harnessing such potential not only requires large investments, but as put forth in this Chapter, coordinated efforts of governments, businesses, and other organisations allied with citizens who accept and support these efforts in their purchasing and voting decisions.

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09

Towards our phosphorus future

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There are abundant opportunities to transition towards more sustainable phosphorus use. Taken collectively, these solutions unlock multiple environmental and societal benefits. Actions must be delivered cooperatively, as part of an integrated plan across sectors and scales. Indeed, coordinated action on phosphorus to support governments, existing conventions, and intergovernmental frameworks, as well as stakeholders, to catalyse improvements in phosphorus sustainability is urgently required. An inter-conventional coordination mechanism to address fragmented phosphorus policy is proposed.

Left: Harmful algal blooms on Lake Erie in August, 2017. The impacts of climate change make such algal blooms more likely in the future. Photo courtesy of Aerial Associates Photography, Inc. by Zachary Haslick. www.flickr.com/photos/noaa_glerl/36546204842

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9.1 Introduction

A decade has passed since the global anthropogenic flow of phosphorus (P) was assessed as having crossed the planetary boundary (Carpenter and Bennett, 2011) - a level of human interference in the global P cycle that results in significant environmental damage. The current societal burden of unsustainable P use is high and getting higher. Nevertheless, as described in the preceding chapters of this report, there are abundant opportunities to transition towards more sustainable P use. Taken collectively, these solutions unlock multiple environmental and societal benefits. Global-scale transition to a sustainable P cycle is essential and possible. By taking the necessary action, humanity can safeguard freshwater ecosystems through emissions reductions and the development of resilient food systems, both of which are fundamental to socio-economic development.

In 2013, the opportunity was highlighted for a 20% improvement in nutrient (P and nitrogen (N)) use efficiency by 2020 across the full chain of food and waste systems (Sutton et al., 2013). Since this time, several nutrient sustainability goals have been proposed. The United Nations Environment Programme (UNEP) Colombo Declaration (UNEP, 2019a) calls for the halving of N waste by 2030, understood as the sum of all N losses (including denitrification of reactive N to N_2). The working group of the Post-2020 Global Biodiversity Framework has proposed to reduce pollution from excess nutrients by 50% by 2030 (CBD, 2020). Similarly, the Farm to Fork strategy underpinning the European Green Deal calls for actions to reduce nutrient losses by at least 50% and fertiliser use by at least 20% by 2030 (European Commission, 2020). It is important to consider the key barriers to such goals.

As is recognised by the preceding chapters, progress on P is not hindered by a lack of scientific evidence on solutions, but instead by a lack of policy and public awareness, a lack of operationalisation of policies, fragmentation of policies, and the absence of intergovernmental coordination.

Signs of progress toward sustainable nutrient use are beginning to appear. The momentum for action on N pollution, for example, is building, with the first United Nations Environment Assembly (UNEA) resolution on Sustainable Nitrogen Management agreed in 2019 (UNEA, 2019a). The Our Phosphorus Future (OPF) project aims to catalyse similar progress for phosphorus. Indeed, the preceding chapters demonstrate that many solutions are within reach for P and would provide multiple benefits across sectors and scales. For example, optimising fertiliser application rates on farms and increasing P recycling reduces losses to fresh waters, which reduces the costs of responding to water quality degradation while supporting green economic growth. Building upon decades of research and innovation across multiple fields, the evidence base is now available with which to re-design the global anthropogenic P cycle. The challenge now is in mobilising governments, industry, stakeholder organisations, academics, and the public to provide leadership in shaping Our Phosphorus Future.

In this final chapter, we draw on the preceding chapters to propose ten key actions towards global P sustainability. We provide a perspective on the opportunities for converting knowledge into international action, noting the lack of inter-governmental attention on sustainable P use. We then identify high-level ambitions to reduce unsustainable P use globally, recognising the need for better integration of P across sustainability policies.

9.2 Ten key actions towards global phosphorus sustainability

We propose ten key actions across sectors that are central to improving sustainable P management globally. The list below is not exhaustive and is framed in a global context. It is important to acknowledge that priorities on actions will differ among regions, where issues may differ as will resources and ambitions to implement solutions.

1. Increase the use of recycled phosphorus in fertiliser and other chemical industries, as an alternative or supplement to phosphate rock.

Economic, legislative and communication instruments are required to support the mineral fertiliser and chemical industries to increase their use of recovered P materials as alternatives to P mined from phosphate rock (PR) (Hermann et al., 2019; Kabbe and Rinck-Pfeiffer, 2019; Matsubae and Webeck, 2019) (see Chapter 7). Common features that will make recovered P materials commercially viable as an industry-compatible raw material include homogeneous quality, low levels of contaminants, and production levels that are high enough to ensure a reliable supply (Schipper, 2019).

2. Optimise phosphorus inputs to agricultural soils and maximise crop uptake to minimise losses.

Fertiliser use can be optimised, as illustrated by the 4R (Johnston and Bruulsema, 2014; The Fertilizer Institute, 2017) and 4R Plus (The Nature Conservancy, 2021) approaches, described in Chapter 4. However, to achieve this, extensive soil P testing is needed. Identification of cost-effective, user-friendly testing methods is therefore critical. In some regions appropriate control limits on P inputs may be required, including consideration of interactions with other nutrient cycles (Masso et al., 2017; Blackwell et al., 2019) (see Chapter 4). In regions with soil P deficiency, P inputs that exceed crop uptake may be required, at least in the short-term (Cordell and White, 2014; Nziguheba et al., 2016) (see Chapters 2, 3 and 4). Soil-crop management to improve crop uptake of applied and legacy P in the soil may embrace integrated soil fertility management, including water and weed management (Blackwell et al., 2010; Vanlauwe et al., 2010), rhizosphere management and the use of P-efficient cultivars and biofertilisers (Shenoy and Kalagudi, 2005; Stutter et al., 2012; Adhya et al., 2015) (see Chapter 4).

3. Optimise animal diets and the use of supplements to reduce phosphorus excretion.

The P content in animal diets can be optimised to match animal growth stage requirements. As commonly practised in most more economically developed countries, supplementing the feed of monogastric animals with phytase enzymes can improve P uptake from grains, thereby reducing P excretion (Ferket et al., 2002; Dao and Schwartz, 2011; Menezes-Blackburn et al., 2013; Oster et al., 2018) (see Chapter 4). Dietary phytase supplements can also increase the bioavailability of P in manures, improving crop uptake when manures are applied to soils thereby reducing the need for mineral P additions (George et al., 2018). However, the impacts of dietary phytase on potential P mobility in the environment and fluxes from soils to waters remain to be explored (see Chapter 4).

4. Increase appropriate application of manures, other phosphorus-rich residues, and recycled fertilisers to soils, to complement appropriate mineral fertiliser use.

Assessments have identified that recycling P, especially in manures, can support a significant reduction in mineral P fertiliser requirements (Vaccari et al., 2019; Powers et al., 2019), and provide additional agronomic benefits, including a source of N, micronutrients, organic carbon and improved soil water retention (Schröder, 2005; Lashermes et al., 2009; Diacono and Montemurro, 2010; Deeks et al., 2013; Pawlett et al., 2015). Phosphorus-rich residues must be appropriately processed to enable storage, transport, avoid decomposition, ensure sanitary safety and supply reliable and consistent agronomic characteristics (see Chapter 6).

5. Improve global reporting and assessments of phosphorus emissions and their impacts on freshwater and coastal ecosystems.

The available evidence on biodiversity loss and ecological sensitivity in response to nutrient pressures is compelling (Darwall et al., 2009; Smith and Schindler, 2009; Chen et al., 2009; Paerl and Paul, 2012) but incomplete. While phosphorus is one of the key drivers of the biodiversity loss emergency in freshwater and coastal ecosystems, the true scale of the problem globally is difficult to estimate as baseline data are lacking in many regions. It has been predicted that 31% of the global landmass contains catchments that may exhibit undesirable levels of algal growth; 76% of which is caused by P-enrichment, affecting 1.7 billion people (McDowell et al., 2020). Emerging evidence suggests that freshwaters may drive globally-significant emissions of greenhouse gases as a result of nutrient enrichment (Beaulieu et al., 2019). Investment in long-term monitoring across a wider range of biomes is necessary to track and study their response to mitigation and nutrient reduction strategies (see Chapter 5). These will provide a critical link between information, evidence-based decision making and policy development and should be used to inform adaptive management frameworks for reversal or halting of biodiversity loss in freshwater and coastal ecosystems.

6. Implement integrated approaches for freshwater and coastal ecosystem restoration and protection at catchment, national and transboundary scales.

Integrated P management strategies that cross scales and work in parallel with efforts to reduce other key nutrient stressors will be essential in achieving improved water quality, tackling biodiversity loss and generating significant socio-economic co-benefits (see Chapter 5). This will require the development of regional targets, mandates and incentives, and transboundary cooperation frameworks to integrate across multiple pillars - from governance to technology, monitoring and assessment - including effective stakeholder engagement (see Chapter 5). Examples of the application of such frameworks leading to fully-costed 'business plans' demonstrate that supporting P flux reductions for restoration makes financial sense (World Bank Group, 2018). However, the complexity of achieving these reductions across sectors must not be underplayed and requires engagement and transparency across both public and private sectors (see Chapter 5).

7. Implement national to global strategies to increase recovery and recycling of phosphorus from solid and liquid residue streams.

Currently, significant amounts of P are lost in 'waste' streams representing solid and liquid residues requiring disposal (Withers et al., 2015a; Vaccari et al., 2019). Multiple strategies exist to improve the recycling of P in manures, abattoir residues, food processing and domestic wastes, sewage derived biosolids and wastewaters (Kabbe and Rinck-Pfeiffer, 2019), all of which represent resources with potential for recycling (see Chapter 6). Such losses not only represent a waste of valuable P but in some cases a significant cause of P pollution to waterbodies (Withers et al., 2015a) (see Chapter 5). National policies that optimise P recycling, and hence reduce reliance on mineral P fertilisers, are acknowledged as pivotal to drive the transition to greater P sustainability (Koppelaar and Weikard, 2013; Cordell and White, 2014; Reijnders, 2014; Withers et al., 2015b, 2018; van Dijk et al., 2016; Chowdhury et al., 2017; van Kernebeek et al., 2018).

8. Ensure sufficient access to affordable phosphorus fertilisers (mineral, organic and recycled) for all farmers.

Improving farmers' access in developing regions to P fertilisers may include better access to credit, extension services, investment in sustainable infrastructure (such as local P recycling systems from food waste and sanitation), and knowledge exchange to support better P use efficiency and recycling (Cordell and White, 2014) (see Chapter 3). Ensuring that all farmers (in particular, small-scale farmers in low-income countries) can access sufficient P to grow crops and are buffered from fertiliser price fluctuations should be developed as a global ambition that will require international cooperation (Nziguheba et al., 2016; Teah and Onuki, 2017) (see Chapters 2 and 3).

9. Promote a global shift to healthy and nutritious diets with low phosphorus footprints.

Wider adoption of healthy diets with low to moderate amounts of meat and dairy could radically reduce demand for mineral P fertilisers (Elser and Bennett, 2011; Vaccari et al., 2019). The biggest gains can be made from reducing meat consumption in countries that already consume more than recommended (WHO, 1999; Metson et al., 2012; Kummu et al., 2017) (see Chapter 8).

10. Reduce the amounts of phosphorus lost as food waste in food processing, retail, and domestic consumption.

Strategies to reduce food waste should be supported by clear evidence on the benefits of change. Most food waste in low-income countries occurs before products reach consumers (see Chapter 8). Wealthier nations waste more food in retail and at home (Kummu et al., 2012). Whilst recycling P from food wastes can contribute to a more sustainable P system, the focus of efforts should be to reduce the production of food waste in the first place (and thereby cutting the resources used to produce it) (Vaccari et al., 2019).

9.3 Supporting International Cooperation

The actions listed above must be delivered cooperatively, as part of an integrated plan across sectors and scales. Whilst national framing of nutrient sustainability strategies is vital, we highlight several P sustainability issues that cannot easily be addressed without international cooperation.

9.3.1 Polices, regulations and initiatives for phosphorus sustainability

Strategies to improve P sustainability are conspicuously absent in most regions (Gross, 2010; Cordell and White, 2015; Rosemarin and Ekane, 2016; Jacobs et al., 2017). Few policies relating to sustainable P management exist at the national scale, and none at the global scale (Chowdhury et al., 2017; Kanter and Brownlie, 2019). Phosphorus was identified a decade ago as an emerging issue in the UNEP Year Book 2011 (Syers et al., 2011). However, relevant international sustainability initiatives rarely mention P, including the Sustainable Development Goals (SDGs) (United Nations Statistical Division, 2016), and the Aichi Biodiversity Targets (CBD, 2020). This is despite the considerable contribution that improving P sustainability can make to achieving their goals and targets (Kanter and Brownlie, 2019).

Strategies that rely on voluntary uptake of P sustainability measures can also be effective. In an assessment of policies and actions to decrease water quality impairment from P, McDowell et al., (2016) showed that a combination of measures was necessary to reduce P pollution in New Zealand, the UK, and the USA. Relevant combinations included mandatory and voluntary measures at the farm to small catchment scale, in combination with policy instruments at larger scales. In some less-economically developed countries, ambitions for increased use of fertilisers to address P deficient soils have yet to be fully achieved. In Africa, the 'Abuja Declaration on Fertiliser for an African Green Revolution' of 2006 called for the elimination of all taxes and tariffs on fertiliser and fertiliser raw material to increase fertiliser use (African Union Special Summit of the African Heads of State and Government, 2006). By 2018, 47 of the African Union states had not yet achieved this, due to a lack of harmonisation on policy and regulation frameworks, lack of tax incentives, trade barriers, and poor quality control on fertilisers (African Development Bank, 2021).

Examples of policies, regulations, volunteer schemes and subsidies related to P sustainability in Australia, Brazil, China, the EU, India, Sub-Saharan Africa (SSA), and the USA are listed in Table 9.1 (see end of chapter), highlighting the high degree of fragmentation in government-supported actions. Experience shows that if policies are to optimise P sustainability, they must adopt a more joined-up approach between water, agricultural and urban planning policy sectors (Tong et al., 2017; Carvalho et al., 2019; Wurtsbaugh et al., 2019). A coordinated policy approach that aims to reduce P loading to water bodies, for example, should be supported by measures to increase P sustainability in agriculture. These must include improved farmer access to recycled P fertilisers, which itself should be supported by policies to encourage P recovery from residue/waste streams and the use of recycled P in the production of fertilisers. In some cases, better enforcement of existing policy measures is also needed, as highlighted by Teenstra et al. (2014) in their analysis of manure management policies in Asia, Africa and Latin America. Furthermore, flexibility should be built into new and existing policies to allow them to adapt to changing conditions, brought about, for example, by climate change, urbanisation, changes in P loading and ecosystem sensitivity to P (Chowdhury et al., 2017; Tong et al., 2017; Duncan et al., 2019).

Policy development and integration are needed across all regions to drive innovation and the behavioural changes needed to deliver P sustainability. To address this, raising government and policymaker awareness of the benefits of sustainable P use is an obvious but critical first step.

9.3.2 Raising awareness of the world's governments

The United Nations Environment Assembly (UNEA) is the world's highest-level decision-making body on the environment, with universal membership of all 193 Member States. A global intergovernmental agreement in the form of a UNEA resolution provides a key opportunity to raise awareness among the world's governments of the importance of P sustainability. UNEA resolutions, whilst not legally binding, represent the joint aspirations of governments, frame consensus around actions to be taken, and help coordinate development aid and technical assistance.

At the fourth session of UNEA (UNEA-4) in 2019, 23 resolutions were adopted (UNEA 2019b). These included resolutions on Sustainable Nitrogen Management (UNEP/EA.4/Res.14), Innovative Pathways to Achieve Sustainable Consumption and Production (UNEP/EA.4/Res.1), Mineral Resource Governance (UNEP/EA.4/Res.19) and Innovations on Biodiversity and Land Degradation (UNEP/EA.4/Res.10) (IISD, 2019). At the fifth session of UNEA (UNEA-5.2) in 2022 (UNEA 2022), a further resolution for Sustainable Nitrogen Management (UNEP/EA.5/Res.2), acknowledged the significant impacts of excessive levels of both N and P in the environment. Additionally, a resolution on Sustainable Lake Management (UNEP/EA.5/Res.4) called for Member States to protect, conserve, and restore, as well as sustainably use lakes. All the resolutions listed above are highly relevant to P sustainability. Development of a future UNEA Resolution on Sustainable P Management would provide a key opportunity to catalyse intergovernmental action on the global P challenge (Brownlie et al., 2021).

There is also a clear opportunity to raise awareness for P through UNEA. For example, the UNEA 4/14 resolution led to the launch of the UN Global Campaign on Sustainable Nitrogen Management, resulting in the development and adoption of the Colombo Declaration on Sustainable Nitrogen Management (UNEP, 2019a).

9.4 Improving governance models to recognise the need for intergovernmental cooperation

As with the experience from the N cycle, policies relevant to P are fragmented between component issues, which can lead to a lack of action and coherency (UNEP 2019d, Sutton et al., 2020; Brownlie et al., 2021). There is a need for coordinated action on P to support governments, existing conventions and intergovernmental frameworks, as well as stakeholders, to catalyse improvements in P sustainability. Possibilities to coordinate N policy engagement more effectively at the national, regional and global levels have been suggested (see Sutton et al., 2020, 2021). Based on these, the following four corresponding options to coordinate policies across the P cycle can be identified:

1. Phosphorus policy fragmentation across policy frameworks (e.g. business as usual).
2. Phosphorus leadership under one existing Multilateral Environmental Agreement (MEA).
3. Establish a new intergovernmental convention to address P sustainability.
4. Establish an 'inter-convention P coordination mechanism' to facilitate cooperation on P sustainability.

Business as usual is not sufficient to address the current P challenge (Carpenter and Bennett, 2011). Similarly, it appears that the mandates of existing MEAs are not broad enough to address the P challenge, providing a barrier to any one of them taking the lead on phosphorus. At present, there appears to be little to no foundation to establish a new intergovernmental convention on P in the near future (5-10 years). Experience with N points to the need to draw as far as possible on the work of existing bodies. This leads the discussion to the fourth option, which can draw on the experience related to the 'Inter-convention Nitrogen Coordination Mechanism' (INCOM), which is currently under development for N (UNEP, 2019b).

Building on this progress (see Sutton et al., 2020, 2021), we identify the need for a complementary 'Inter-convention Phosphorus Coordination Mechanism' (PHOSCOM), where lessons from the experience for N can also inform the possible relationships (Figure 9.1). For example, under the current development for INCOM, the International Nitrogen Management System (INMS) could be embedded as a science support process under the overarching inter-governmental mandate of INCOM. In the same way, an emerging concept for PHOSCOM could embed scientific assessments to outline risks and identify adaptation and mitigation options.

An obvious question that needs to be considered is the relationship between the developing concepts for INCOM and PHOSCOM. This may be considered especially as a matter of timing and communication, noting that the issue of 'phosphorus hypersensitivity' can provide a barrier to progress on sustainable P management (Chapter 2). Care is needed to avoid that such specific barriers for P do not hold up progress on interconvention coordination on nitrogen. However, there is a need to further examine where policies for N and P can be aligned to maximise synergies (Kanter and Brownlie, 2019). While there are generic lessons, substantial differences between issues and the component interlinkages for P and N exist. For these reasons, it is acknowledged that at present there is an operational benefit in developing the processes in parallel, rather than a simple combination. However, it is also obvious that there is potential for sharing of lessons learned and for cooperation between the P and N communities, for example, as part of the UNEP Pollution Implementation Plan, which also makes links to the Basel Rotterdam and Stockholm (BRS) Conventions on toxic substances and to ongoing actions on plastic pollution (UNEP, 2018, 2019c). While recognizing the need to further develop such higher-level cooperation, we focus the following comments on the immediate challenges for phosphorus.

One of the key points to recognise in addressing international cooperation on P and N is the distinction between intergovernmental action and multi-actor engagement. In this regard, INCOM is developing as an intergovernmental process, following up UNEA Resolution 4/14, as illustrated by the agreement of member states under the Colombo Declaration (UNEP, 2019a). Conversely, UNEP established the Global Partnership on Nutrient Management (GPNM) in 2009, which represents a multi-actor partnership linking business, academia, civil society and governments among others. In the same way, substantial connections are relevant with the UNEP Global Wastewater Initiative (GW2I), with both these bodies established under the Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA). These bodies have an important role to play in catalysing improved understanding between different actor groups across society. In this way, the parallel establishment of INCOM and PHOSCOM provides opportunities to increase the focus of intergovernmental action on both N and P, which in return may strengthen multi-actor engagement in both GPNM and GW2I as a basis to address barriers and accelerate change.

Figure 9.1 indicates potential intergovernmental contributions in delivering a joined-up approach to P management. The starting point is engagement by UN Member States. Knowledge exchange between Member States and the various intergovernmental bodies highlighted in Figure 9.1 would help guide specific actions on P sustainability. Such an intergovernmental coordinated approach is also necessary to develop a shared ambition by governments internationally (see Chapter 1).

Although national scale strategies to improve P sustainability are critical, a further challenge is that several P sustainability issues will be difficult to address without international cooperation. These include, for example:

- Displacement of ecosystem impacts of production (Nesme, 2016; Hamilton et al., 2018; Li et al., 2019).
- Phosphorus pollution of transboundary/shared waters (see Chapter 5; Bohman, 2017; Jetoo, 2018).
- Trade of contaminants in PR and P fertiliser (see Chapters 2 and 7).
- Creation of global markets for recycled P materials and products (see Chapter 7).
- Ensuring all farmers have sufficient access to P to grow crops (see Chapter 3).
- Responding to PR/mineral P fertiliser price volatility (see Chapter 2).

The breadth of these issues highlights the need for cooperation across the environment, through the agri-food chain, and embracing the development of the circular economy and engaging with issues of P trade.

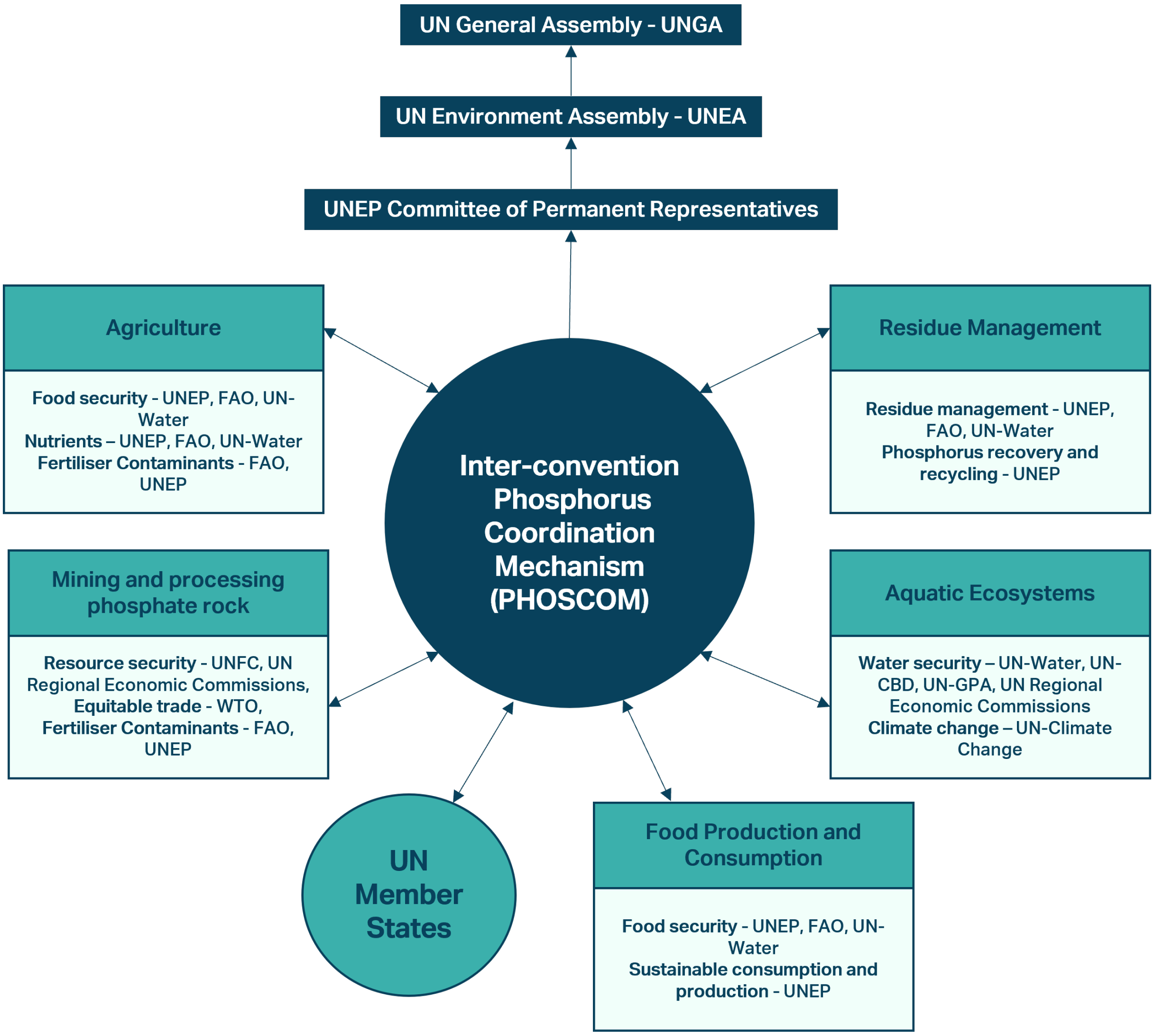

Figure 9.1 There is currently little policy coordination on phosphorus (P) across scales. Building on progress made in the nitrogen community (Sutton et al., 2020, 2021; UNEP 2019c), an 'Inter-convention Phosphorus Coordination Mechanism (PHOSCOM) is proposed to link P science-policy support among existing intergovernmental frameworks and other initiatives, under the auspices of the United Nations Environment Programme (UNEP). Key bodies with relevant interests include UNEP, the Food and Agriculture Organization of the United Nations (FAO), UN-Water (including the UN Convention on the Protection and Use of Transboundary Watercourses and International Lakes (UN-WC)), Regional Seas Conventions (RSCs) and the UN Convention on the Law of the Sea (UNCLOS)), the UN Regional Economic Commissions (which includes the UN Economic Commission for Europe (UNECE) Convention on Access to Information (Aarhus Convention), and the UNECE Water Convention (UNECE-WC)), the UN Framework Classification for Resources (UNFC), the UN Resource Management System and its regional initiatives, the World Trade Organization (WTO), the UN Convention on Biological Diversity (CBD) and the UN Climate Change Convention (UN-Climate Change) and regional/ national processes.

9.5 An aspirational goal for phosphorus sustainability

Time-bound goals have a critical role to play in developing environmental policy, serving as a 'starting gun' for action, and a 'finishing line' for delivery. To propose a meaningful aspirational goal for P, it is therefore useful to identify a potential finishing line for global P sustainability. In this regard, achieving 'P security' may be a suitable goal. This has been defined by Cordell (2010) as "…all the world's farmers have access to sufficient P in the short and long term to grow enough food to feed a growing world population, while ensuring farmer livelihoods and minimising detrimental environmental and social impacts". Achieving such a goal will not be possible without significant progress towards a 'closed human P cycle' (Childers et al., 2011). Opportunities to make greater progress towards a circular P system have been identified in the preceding chapters of this report.

Whilst closing the P cycle does not address P access issues directly, it is widely acknowledged in the literature as an overarching aim of strategies to improve P sustainability (Elser and Bennett, 2011; Childers et al., 2011; Bateman et al., 2011; Cordell and White, 2014; Shepherd et al., 2016; Scholz and Wellmer, 2018; Withers et al., 2018). The question then becomes, what goal will help make progress towards closing the global P cycle and is it appropriate across all regions?

A goal to reduce the rate of P mining or the use of mineral P in fertiliser production does not directly address P losses that cause pollution (e.g. those from inefficient agricultural management or sanitation). Furthermore, without the supporting policies and infrastructure in place to deliver large scale P recycling, food security could be compromised. This highlights the need to focus on mobilising increased P recycling, for which a goal for P-containing fertiliser products to contain a minimum fraction of 20% recycled P by 2030 was identified in Chapter 2.

A goal for improving P use efficiency is appropriate for all regions, recognising that this may not translate into improved yield in some places. Sutton et al. (2013) provide a strong argument for improvement to nutrient use efficiency (NUE) across the food-value chain (i.e. a 20% improvement in full chain NUE, by 2020, from 2008 levels) as a shared goal. However, whilst this may apply to N, in regions such as Sub-Saharan Africa where soils are P depleted, low agricultural PUE may be arguably desirable, provided excess P is stored in the soil, rather than exported from soil to water (Langhans et al., 2021). Such a long-term accumulation of P in P deficient agricultural soils could be considered an investment in the future (Syers et al., 2008; Menezes-Blackburn et al., 2018). Furthermore, using a relative improvement for use efficiency (i.e. 20% of NUE) implies those countries with the highest efficiency would need to improve more than those with low efficiency.

A high-level goal that directly addresses the reduction in P pollution and supports an increase in P recycling may therefore be most appropriate. Such a goal should be applicable to all countries, covering all sectors, and not adversely impact access to P fertilisers. Bearing in mind the necessity of sufficient time to mobilise change, we, therefore, propose **an aspirational goal of a 50% reduction in global P pollution and a 50% increase in the recycling of P lost in residues/wastes, by 2050. We term this the '50:50:50' aspirational goal.**

9.5.1 Unpacking the 50:50:50 goal

Achieving the 50:50:50 goal would reduce impacts to P polluted ecosystems (as outlined in Chapter 5), whilst still delivering enhanced resilience to the global food system (as outlined in Chapter 3) with far-reaching benefits across the seven OPF pillars (Figure 9.2). Whilst the strategies to achieve this goal may differ widely among nations, its overarching aims, to reduce P pollution and increase food system resilience, remain universally relevant.

Based on current estimates of global anthropogenic P loading to water bodies, a 50% reduction in global P pollution will require a reduction in anthropogenic P loading to water bodies by 3.1–7.0 Mt P year^{-1} (Chen and Graedel, 2016; Beusen et al., 2016). A 50% reduction in total annual P loading to surface waters is proportionately equivalent to 2.5–6.2 Mt P year^{-1} from agriculture alone (Chen and Graedel, 2016; Beusen et al., 2016). Opportunities to reduce P losses to water from agriculture are highlighted in Chapter 4. Assuming losses of P from agricultural systems are subsequently replaced with inputs of mineral P fertiliser, a 50% reduction in P pollution from agriculture could save the global farming community between $US8.0–19.8 billion year^{-1} in mineral fertiliser costs. This is based on a cost of P in diammonium phosphate (DAP) of US$3.2 P kg^{-1} (for September 2021)[i], and assumes all losses are replaced by DAP. With substantial fluctuations in DAP price (e.g. ranging from US$280-643 DAP t^{-1} between 2010 and 2021) this value varies greatly. Based on the average DAP price between 2010 to 2021 the annual global value of the saving would be US$5.7-14.1 billion.

A 50% reduction in P pollution would have a significant impact on the costs associated with the management of P loading to water bodies. As discussed in Chapter 5, the most effective and least expensive strategy is to stop P pollution at its source (e.g. improving fertiliser management, improving sanitation). The second is to implement adaptation interventions in the landscape to intercept P flows from reaching water bodies (e.g. buffer strips, erosion control). The median estimated cost for P pollution adaptation interventions is US$46 kg^{-1} of P intercepted (see Chapter 5). A 50% reduction in P pollution could therefore reduce the requirement for annual adaptation costs[ii] by between US$143–322 billion.

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[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. Based on the cost of DAP in Sept 2021 (US$643 t^{-1}), 4.98 t of DAP would contain 1.0 t of P, and would cost USD$3203.

[ii] Costs are calculated by multiplying the estimated range of P pollution by interventions costs (see Chapter 5). A 50% reduction in global P pollution will require a reduction in anthropogenic P loading to water bodies by 3.1–7.0 Mt P year^{-1}, whilst adaptation interventions are estimated at US$46 kg^{-1} P intercepted (see Chapter 5).

Where the preceding steps cannot be achieved, measures to capture the P within the waters can be implemented. A 50% reduction in pollution would slow the increase in global P stockpiles retained in freshwater ecosystems, currently increasing by 5 Mt year^{-1} (Beusen et al., 2016). Geoengineering techniques as described in Spears et al. (2013), to control the additional P being loaded into these stockpiles (e.g. make it not bioavailable) would cost US$122–220 billion year^{-1} (see Chapter 5). A 50% reduction in P pollution could therefore reduce the requirement for additional geoengineering costs by US$61–110 billion year^{-1}. Whilst such estimates are difficult to make with accuracy, recovery of waterbodies at this scale is unlikely to be implemented due to high costs and, instead, the cost of damaged ecosystems are paid by society in terms of losses in ecosystem service.

A 50% reduction in P pollution would have a significant impact on the economic losses associated with eutrophication impacts. As examined in Chapter 5, quantitative data on these costs is lacking in the literature. Of the few studies published, the costs of eutrophication of fresh waters in the US was estimated at US$2.2 billion annually (covering losses to industry, real estate, and management for conservation of endangered species and drinking water supply) (Dodds et al., 2009). Similarly, for England and Wales in the UK, losses as a result of eutrophication of fresh waters were estimated at £75–114 million year^{-1} (US$104–158 million year^{-1}) (covering damages and management interventions), with an additional £55 million year^{-1} to cover the policy response (Pretty et al., 2003). A similar assessment updates this figure, estimating eutrophication in the UK in 2018, to cost £173 million (US$220 million) (Jones et al., 2020). We note, for some polluted lakes, a 50% reduction in P pollution may not be sufficient for ecological recovery, whilst for pristine lakes, a 50% reduction may not be required. Targeting of action to reduce P pollution should be implemented through national strategies, to identify where reductions will make the most impact.

In addition to the direct costs of P driven water pollution, recent reports indicate that P-enriched waterbodies are also an important source of greenhouse gas emissions (Beaulieu et al., 2019). Downing et al. (2021) estimate the global social cost of eutrophication-driven methane emissions from lakes between 2015 and 2050 to be $7.5–$81 trillion. These values warrant further investigation and critical analysis, given the very large estimated costs.

A 50% increase in the recycling of P lost in organic waste streams represents an increase in 8.5 Mt P returned to food production annually (based on data presented in Chen and Graedel, 2016).[i] This could offset US$27 billion year^{-1} as spent by farmers on mineral P fertilisers (based on a DAP price in September 2021 of US$643 t^{-1} - see footnote on page 362). By increasing the conversion of P-rich organic wastes into value-added products (e.g. fertilisers), waste-producing industries may also avoid/offset waste disposal costs (Peccia and Westerhoff, 2015).

[i] This is composed of a 50% increase in the recycling of crop residues by 1.3 Mt P year^{-1}, animal manures by 2.2 Mt P year^{-1}, food processing wastes by 1.6 Mt P year^{-1}, human excreta and other human wastes by 3.5 Mt P year^{-1}, based on the analysis of global P flows in 2013, as presented in Chen and Graedel (2016).

Overall, considering these potential savings, a global total of US$150-369 billion year^{-1} of resource-saving would be provided by the 50:50:50 goal,[i] which could stimulate the circular economy for sustainable P management. This does not include savings made through reductions in economic losses associated with eutrophication impacts or eutrophication-driven methane emissions. The financial value of these benefits provides the opportunity to mobilise action on processing, storing, and transporting P-rich organic materials, where financial synergies may be found in simultaneously mobilising recovery of N and other elements.

Besides the agronomical benefits of recycling organic residues (see Chapter 6), short-term benefits may also include a potential reduction of P losses to waterbodies (assuming P-rich organic materials are sustainably managed) (see Chapter 5), and reduction in the externalities associated with mining PR and the manufacture of mineral P fertilisers (see Chapter 2; World Bank, 2007; U.S. Environmental Protection Agency, 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).

There are also benefits for food system resilience. A recent study by Vaccari et al. (2019) demonstrated if global P losses from agricultural land, manure use, and food waste were reduced by 50%, and recycling of P from food and human wastes were simultaneously increased (by factors of 5.5 and 3.3, respectively), then enough P would be provided to sustain over four times the current population. Additionally, if the consumption of animal products was also reduced by 50%, almost six times the global population could be sustained (Vaccari et al., 2019).

The '50:50:50' goal aligns with several aspirational goals that have also called for reductions in nutrient losses in recent years. In 2018, the INI announced its commitment to halve N waste by 2030, at the Our Ocean Conference in Bali (https://ourocean2018.org/). This was adopted as an ambition by a group of UN Member States in the Colombo Declaration in 2020 (UNEP, 2019a). As mentioned above this concept has also been embraced by the EU Farm to Fork strategy (European Commission, 2020) and in proposals from the working group of the Post-2020 Global Biodiversity Framework to reduce pollution from excess nutrients by 50% by 2030 (CBD, 2020). Whilst it can be argued these messages carry a high degree of subjectivity (e.g. the definition of loss and pollution), the priority of these messages is to be easily understood and to provide direction (i.e. pollution reduction, recycling increase). Importantly, all these goals identify that those that cause the greatest pollution/losses have the most work to do.

The '50:50:50' goal aligns with existing international initiatives, such as the targets of the SDGs, which may help to initialise the coordinated actions that will be needed. The goals mentioned above call for action by 2030. The '50:50:50' goal can thus be considered as a starting point for discussion by governments and stakeholders across society concerning both the ambition level and urgency in relation to dates for other international goals.

[i] This figure combines estimated costs benefits of a reduction in 50% losses from agriculture which would be subsequently replaced with inputs of mineral P fertiliser (up to 19.8 billion year^{-1}), the increase of 8.5 Mt P returned to agriculture through a 50% increase in recycling, which could offset US$27 billion spent by farmers on mineral P fertilisers, and a reduction in the requirement for eutrophication adaptation costs of up to 322 billion year^{-1}.

THE OPF '50:50:50' GOAL

Calls for a 50% reduction in global phosphorus pollution and a 50% increase in the recycling of phosphorus lost in wastes, by 2050, could deliver benefits across all the OPF pillars.

PHOSPHORUS ACCESS

The 50:50:50 goal could return an additional 8.5 Mt of recycled phosphorus to farms each year, supporting food production and food system resilience.

FOOD SECURITY

The 50:50:50 goal could help create a food system that could provide enough phosphorus to sustain over 4 times the current population.

AGRICULTURE AND FOOD PRODUCTION

The 50:50:50 goal could save global farmers almost $US20 billion in annual mineral phosphorus fertiliser costs needed to replace losses.

WATER QUALITY

The 50:50:50 goal could significantly reduce the impacts of eutrophication, cutting the need for adaptation costs by over US$300 billion a year, with multiple socio-economic benefits of restored ecosystems.

PHOSPHORUS RECYCLING

The 50:50:50 goal could help a transition to a circular economy for the phosphorus cycle; decoupling economic growth from the consumption of finite phosphate rock resources.

PHOSPHORUS RECOVERY

The 50:50:50 goal could help develop sustainable business opportunities, accelerating and supporting new jobs through emerging green economy sectors.

CONSUMPTION

The 50:50:50 goal could provide consumers with better access to foods produced in phosphorus sustainable ways, allowing consumers to better support a transition to a sustainable phosphorus future, sustainable city living and post COVID-19 'Green Recovery'.

Figure 9.2 Key benefits to the environment and society of delivering a 50% reduction in global phosphorus (P) pollution and a 50% increase in the recycling of P lost in wastes, by 2050 (as outlined by the OPF 50:50:50 goal). Benefits are observed across all seven of the OPF pillars.

For example, governments might wish to align with a more urgent ambition for 2030, which would be technically feasible and consistent with other initiatives (e.g. UNEP, 2019c; CBD, 2020; European Commission, 2020). Conversely, it is recognised that there are substantial barriers to change for P, including barriers associated with P hypersensitivity (Chapter 2), while timescales of restoration of lakes laden with excess P can take decades (Chapter 5). For these reasons, we here focus the goal on 2050, not only because there is also still significant work to do to get to the starting line but also because policymaker and public awareness of P issues (in comparison to N) are generally low, and work is needed to catch up.

Finally, this goal is not designed to supersede existing legislation on meeting environmental quality standards for P in water bodies, as covered in the assessment of SDG 6.3.2 (UNEP, 2021). Instead encourage actions towards meeting such targets, or implementing them, where they do not already exist.

We note, in addressing this goal, there are considerable opportunities to align with the ambitions for other nutrient cycles, especially N (Sutton et al., 2013; Kanter and Brownlie, 2019). There is also a need to do so to avoid pollutant swapping between nutrient forms and environmental domains. Whilst the global cost of nutrient pollution has not yet been comprehensively assessed and will inevitably be subject to many uncertainties and debate, global annual societal costs of N pollution have been estimated at between US$ 200-2000 billion (Sutton et al., 2013).

9.6 The next steps for delivering a sustainable phosphorus future

Ten years have passed since UNEP acknowledged a critical need to identify a route-map for sustainable P management, built on consensus among stakeholders at national to global scales (Syers et al., 2011). Here we propose the next steps to build on the efforts of past and existing projects, platforms, conferences, and publications that have developed the field of global P sustainability (Figure 9.3). We expect that the next steps will require a simultaneous top-down and bottom-up approach.

The top-down approach aims to see governments agreeing that addressing P sustainability is a critical issue and ultimately agreeing to goals for improvement. A major part in the success of this step will be to capitalise on the momentum already established by past efforts and to ensure the publication of this report and its associated media continues to raise public and policymaker awareness of the opportunities to improve P sustainability. As noted, this could be recognised in the form of a UNEA resolution.

The bottom-up approach aims to establish a framework to exchange and consolidate the evidence from within the P community to develop the strategies to deliver on-the-ground improvements to deliver the goals.

Building on such actions, work will be needed to convert the aspirational 50:50:50 goal into more specific targets to be achieved. This would set the opportunity to inform national legislative actions by each country/government, which would account for the highly diverse and regional context of nutrient issues and can be supported by the mobilisation of evidence and awareness through the proposed PHOSCOM approach (Figure 9.1).

The steps proposed here have been informed by the progress made by the international N community over the last decade (see UNEP 2019d, Sutton et al., 2020, 2021). This includes the development of a joined-up approach to N management through the integration of international knowledge facilitated by the 'Towards an International Nitrogen Management System project' (INMS) (www.inms.international/), with current work looking to embed this within the intergovernmental INCOM approach. Such a science-based approach is also needed for phosphorus. For example, developing aspirational goals for a UNEA resolution on P will greatly benefit from data outputs of a project to establish an International Framework for Phosphorus, the development of which can be informed by lessons learned from INMS. A science-based international framework for P could play an instrumental role in establishing a PHOSCOM, as many of the bodies involved in a potential PHOSCOM may work together as partners within such a framework. For example, further analysis will be needed to respond to concerns over slow progress made to date, as highlighted in the Helsinki Declaration.[i] What is abundantly evident is the present lack of coordinated intergovernmental action on P urgently needs to be addressed. The ideas expressed here should be seen as a starting point to mobilise actions by UN Member States, and actors across society, all of whom stand to gain from sustainable P management for multiple environmental, health, climate and economic benefits.

Action is needed now to ensure the global 50:50:50 goal can be reached. Unlike actions to address carbon emissions, the benefits of implementing measures to improve P sustainability will be observed mostly at local to national scales. Whilst some countries may not be able to take immediate action, any delay will only accrue further impacts and societal costs.

[i] In 2019 over 500 scientists signed 'The Helsinki Declaration' calling for transformation across food, agriculture, waste and other sectors to deliver much-needed improvements to global P sustainability - mobilised through intergovernmental action. The Helsinki Declaration is available at www.opfglobal.com

Figure 9.3 Steps proposed to consolidate the knowledge, identify opportunities, and catalyse the political support needed to achieve significant improvements in phosphorus sustainability over the next decade.

Table 9.1 Examples of relevant policy/regulations, volunteer schemes and subsidies that have considered phosphorus (P) sustainability, and key references that discuss policy and regulations regarding P sustainability.

Country /region	Policy/regulations specifically designed to address P sustainability	Policy/regulations that contain measures that address P sustainability	Policy/regulations that support measures that address P sustainability, but do not mention P directly	Volunteer schemes and subsidies to address P sustainability	Key References
Australia	None	Nutrient discharge limits (only for wastewater treatment plants in sensitive catchments).	Unclear	The 'FertCare®' program; a voluntary industry mechanism led by Fertiliser Australia (industry association); consistent with the 4Rs. Limited guidance on P discharge provided for private sewage works/septic tank owners provided by the government.	Agriculture and Resource Management Council of Australia and New Zealand and the Australian and New Zealand Environment and Conservation Council (1997) Cordell et al. (2013)
Brazil	None	Some local environmental laws limit organic residue application based on P content and soil properties (not well documented).	Unclear	Industrial practices for P recycling can apply for special financial support from governmental agencies. Some governmental programmes exist such as the 'ABC' programme (a low carbon agriculture programme) where farmers can receive subsidies for better agricultural practices.	Ministerio Do Meio Ambiente (2016) Marques de Magalhães and Lunas Lima (2014)

Country /region	Policy/regulations specifically designed to address P sustainability	Policy/regulations that contain measures that address P sustainability	Policy/regulations that support measures that address P sustainability, but do not mention P directly	Volunteer schemes and subsidies to address P sustainability	Key References
China	None	In 2015, the Zero Increase Action Plan was announced by the Ministry of Agriculture (MOA). This plan required the annual increase in total chemical fertiliser use to be less than 1% from 2015 to 2019 without any yield losses and with no further increases from 2020. China's Water Pollution Prevention and Control Action Plan ("10-Point Water Plan"), and national wastewater discharge standards and pollutant cap-control targets address nutrient pollution. The Ministry of Ecology and Environment (MEE) of China issued a notice on strengthening P management of point source pollution in 2018. The notice requires that key industries establish their baseline of P emissions.	Unclear	In 2005, the technology of soil-testing and fertiliser recommendation (STFR) was strongly advocated by the Chinese government as a national action. To encourage farmers to adopt STFR, the Chinese government committed RMB700 billion in subsidies to provide technical service on soil testing for 190 million farm households, covering 80 million hectares of arable land.	Standing Committee of the National People's Congress (2008) Ministry of Environmental Protection of the People's Republic of China (2012, 2014) MOA (2015) National Health and Family Planning Commission of the People's Republic of China (2015) Liu et al. (2016) MEE (2018) Jiao et al. (2018) Wang et al. (2018)

Country /region	Policy/regulations specifically designed to address P sustainability	Policy/regulations that contain measures that address P sustainability	Policy/regulations that support measures that address P sustainability, but do not mention P directly	Volunteer schemes and subsidies to address P sustainability	Key References
EU	The EU Critical Raw Materials List (contains PR and white P). The EU Fertilising Products Regulation (EU) 2019/1009, opens the market for recycled P and sets cadmium limits for phosphate fertilisers. Several national policies address P recovery from wastewaters (e.g. in Germany, Switzerland) and the Baltic Sea Region (HELCOM) and limit P inputs to crop- or grassland (e.g. Belgium, the Netherlands).	Nitrates Directive, Urban Wastewater Treatment Directive, Water Framework Directive, all address water quality issues (e.g. related to nutrient pollution).	Common Agricultural Policy, Industrial Emissions Directive (e.g. Best Available Techniques (BAT) for large poultry and pig farms). The EU Environmental Action Plan and national environmental legislation, EU and national soils policies, pharma-ceuticals and microplastics policies and landfill taxes (impact sewage biosolids valorisation), and numerous EU and national Circular Economy policies.	National and regional support schemes in place to reduce land degradation (e.g. Austria's ÖPUL).	Hermann and Hermann (2019) Kabbe et al. (2018)

Country /region	Policy/regulations specifically designed to address P sustainability	Policy/regulations that contain measures that address P sustainability	Policy/regulations that support measures that address P sustainability, but do not mention P directly	Volunteer schemes and subsidies to address P sustainability	Key References
India	None	National Food Security Mission; Fertiliser subsidy schemes.	The 'Doubling farmers income' policy and Vision Ganga 2017 (which has provisions for the circular economy through recycling of nutrients from wastewater).	Soil Health Card scheme by the Government of India, which provides farm-level fertiliser recommendations based on soil nutrition and the types of crops grown. The Government of India's Ministry of Agriculture and Farmers Welfare promotes the Small Farmers' Agri-business Consortium (SFAC), with a mandate to not just raise awareness but also encourage the formation and growth of Farmer Producer Organisations (FPOs) http://sfacindia.com/, including efficient use of fertilisers.	Srinivasarao et al. (2013) NGBRA (2017) Chand (2017) Department of Agriculture, Cooperation & Farmers Welfare (2008) National Mission for Sustainable Agriculture (2015) cGanga & NMCG (2019)
New Zealand	None	Nutrient discharge limits for streams and rivers, as related to dissolved P and periphyton growth, and total P to limit phytoplankton growth in lakes. Guidance on default guideline values for further investigation of P contamination of freshwaters.	Unclear	Mandatory farm plans to reduce loss of P from land to water. Fertmark and FertSpread schemes consistent with the 4Rs.	ANZG (2018) McDowell et al. (2013) New Zealand Government (2020) Office of the Minister for the Environment and Office of the Minister of Agriculture (2020)

Country /region	Policy/regulations specifically designed to address P sustainability	Policy/regulations that contain measures that address P sustainability	Policy/regulations that support measures that address P sustainability, but do not mention P directly	Volunteer schemes and subsidies to address P sustainability	Key References
Sub-Saharan Africa	The 'Abuja Declaration on Fertiliser for an African Green Revolution' (2006) calls for the elimination of all taxes and tariffs on fertiliser and to increase fertiliser use.	Unclear	Unclear	Fertiliser subsidies were meant to improve access to the input, but remain controversial, with limited success (e.g. Malawi).	Chirwa and Dorward (2013) Jayne and Rashid (2013) African Development Bank (2021)
USA	None	Clean Water Act (33 U.S.C. §1251 et seq.) almost exclusively regulates point-source pollutants. States have varying levels of regulations on P from both point and non-point-sources. For non-point sources, some states require nutrient management plans from farms, sometimes tied to a P-index.	Food waste recovery mandates are being instituted by many cities, including San Francisco and New York.	Farmers are incentivised to adopt conservation practices through grant programs, most notably the USDA's Environmental Quality Incentives Program (EQIP). There are over 20 such USDA programs. Water Quality Trading Programs have emerged in dozens of watersheds that do/will permit trading of rights to pollute between and among point sources and non-point sources of nutrients.	Breetz et al. (2004) Stubbs (2010)
World	None at the global scale	None at the global scale	None at the global scale	The United Nations 'Sustainable Development Goals' provide a range of ambitious goals from improved water quality to reduced food waste, but do not mention P directly in targets.	United Nations (2014) Kanter and Brownlie (2019)

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Our Phosphorus Future

The challenges and solutions to improving phosphorus sustainability

Prepared by the Our Phosphorus Future network with support from the NERC International Opportunities Fund, the United Nations Environment Programme and the Global Environment Facility through the 'International Nitrogen Management System', and the European Sustainable Phosphorus Platform.

This report draws attention to the multiple benefits and threats of human phosphorus use. It highlights the critical role phosphorus plays in food security and bioenergy needs, as an essential component of fertiliser. Whilst geological depletion of phosphate rock reserves is not an immediate threat, geopolitical, institutional, economic, and managerial factors may impact phosphorus access. It demonstrates phosphorus emissions to water bodies are a key driver of the biodiversity loss emergency in aquatic ecosystems. Impacts include toxic algal blooms, mass fish kills, greenhouse gas emissions, and the loss of economic, societal, and cultural value associated with high-quality ecosystems.

Ten key actions are identified that would help maximise the benefits of phosphorus use for humanity, whilst minimising environmental threats. The actions listed should be delivered cooperatively, as part of an integrated plan across sectors and scales. Whilst national framing of nutrient sustainability strategies is vital, several phosphorus sustainability issues cannot easily be addressed without international cooperation. Examples of current national and regional phosphorus policies are illustrated, revealing green shoots of success in the transition to phosphorus sustainability. However, coordinated action on phosphorus to support governments, existing conventions, and intergovernmental frameworks, as well as stakeholders, to catalyse improvements in phosphorus sustainability is urgently required.

The report highlights there is a need for an inter-conventional coordination mechanism to address fragmented phosphorus policy. A blueprint for such an initiative is outlined, considering institutional options. The potential for economic benefits of improving phosphorus sustainability is illustrated by estimating the consequences of meeting a global aspirational goal to make a 50% reduction in global phosphorus pollution and a 50% increase in the recycling of phosphorus lost in residues/wastes, by 2050. The ideas expressed in this report should be seen as a starting point to mobilise actions by UN Member States, and actors across society, all of whom stand to gain from sustainable phosphorus management for multiple environmental, health, climate, and economic benefits.

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