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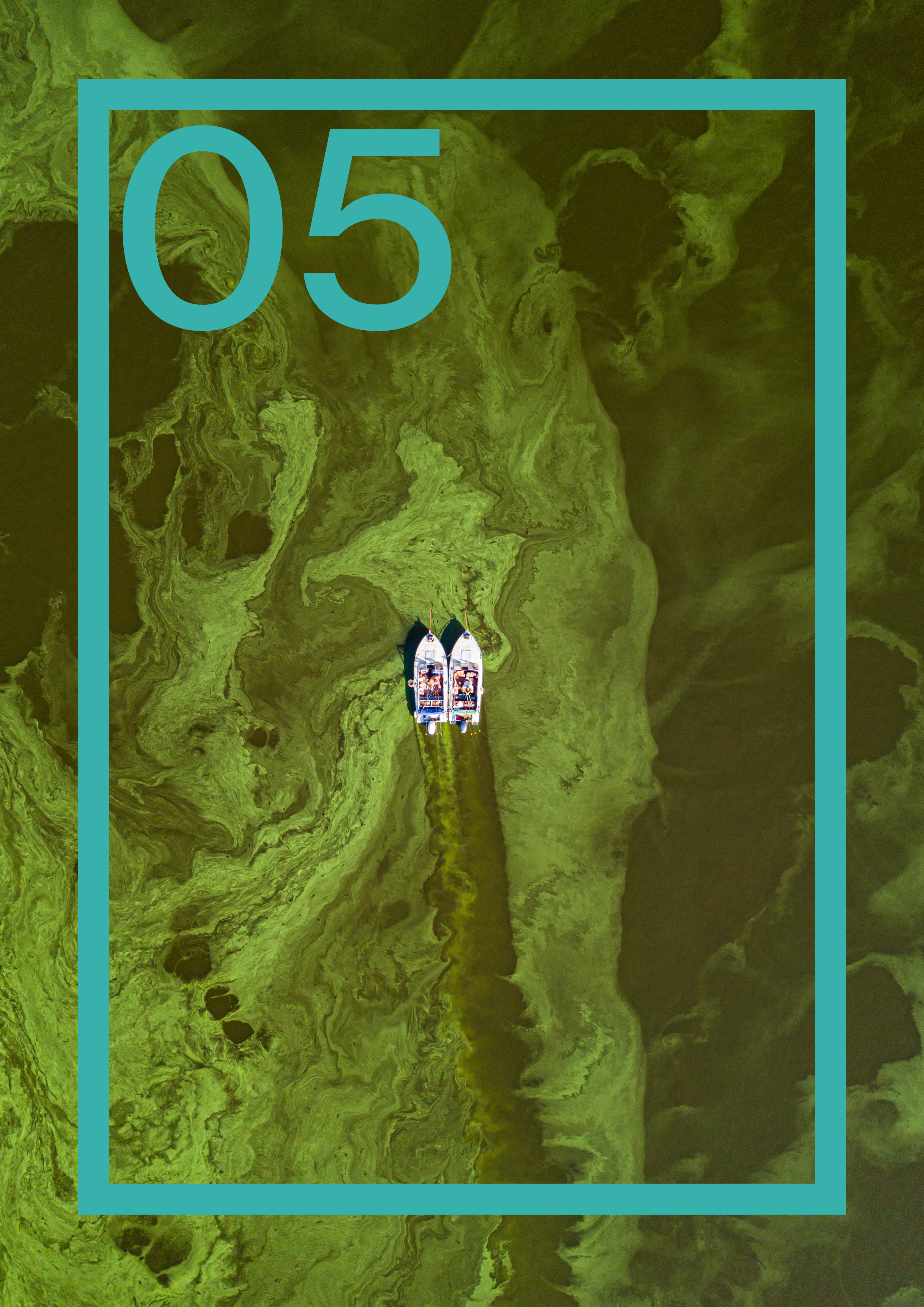

Phosphorus and water quality

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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.

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

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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

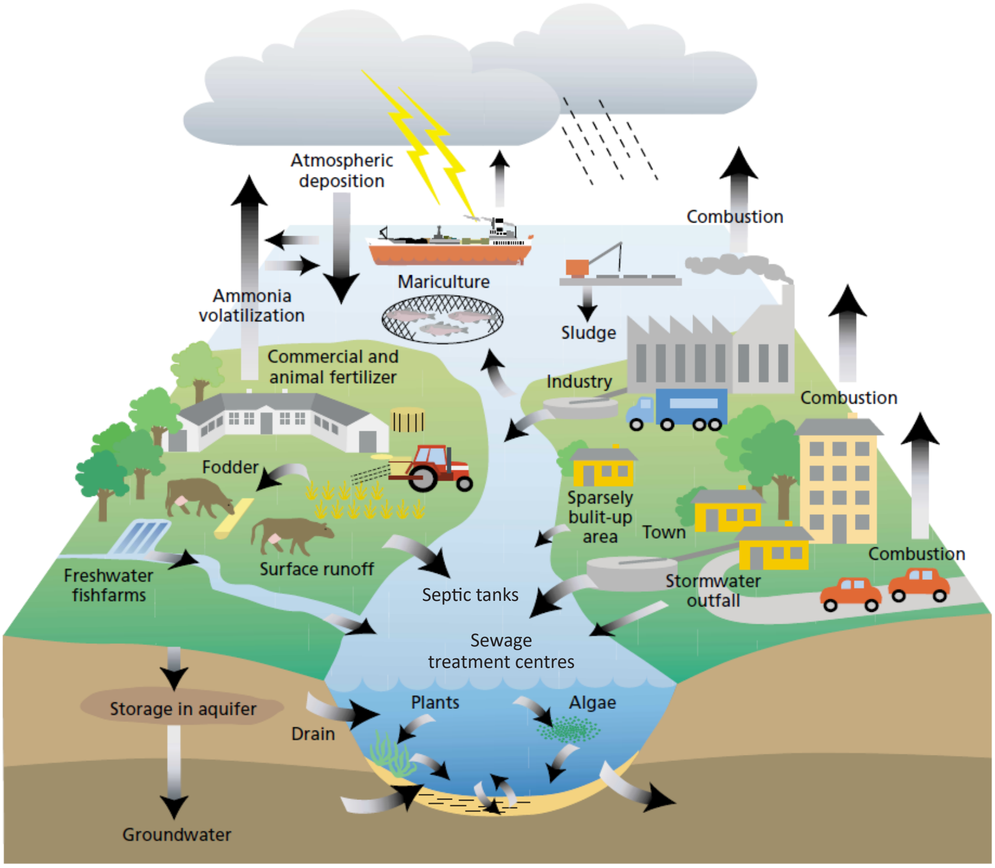

Figure 5.1 Anthropogenic sources of phosphorus to the aquatic environment. The main anthropogenic nutrient sources are discharges from agricultural runoff, sewage treatment works and industry. Modified from Ærtebjerg et al. (2003).

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).


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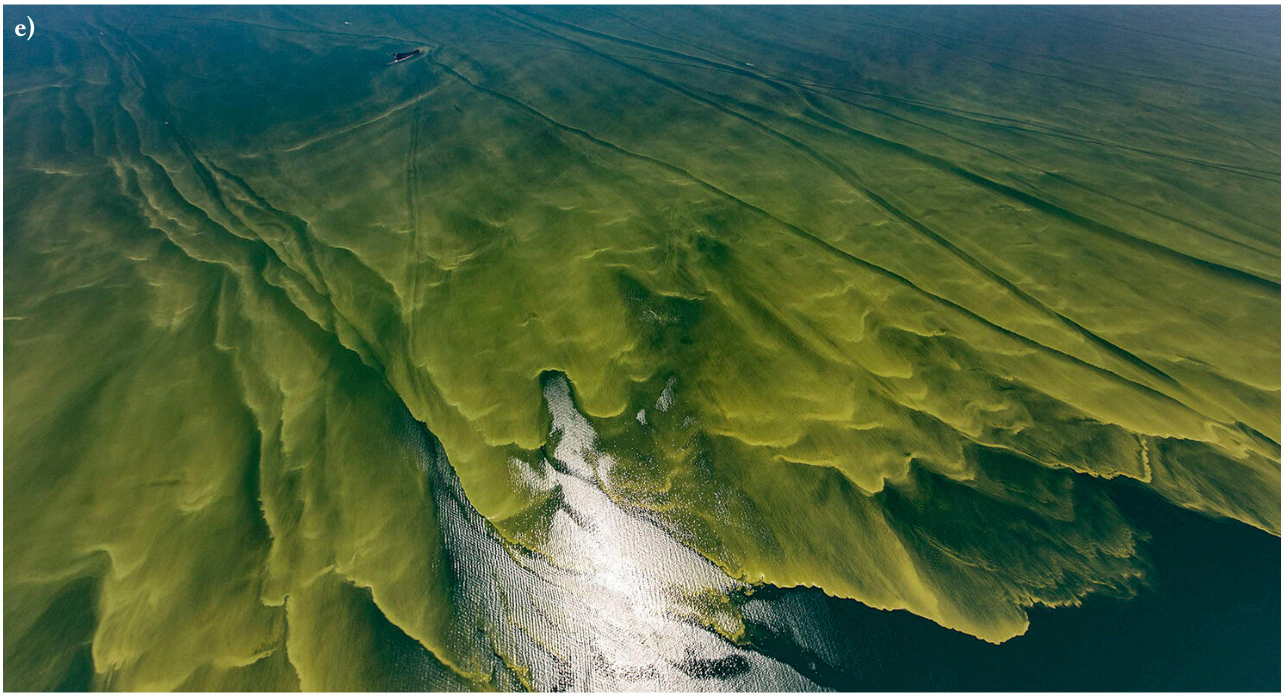

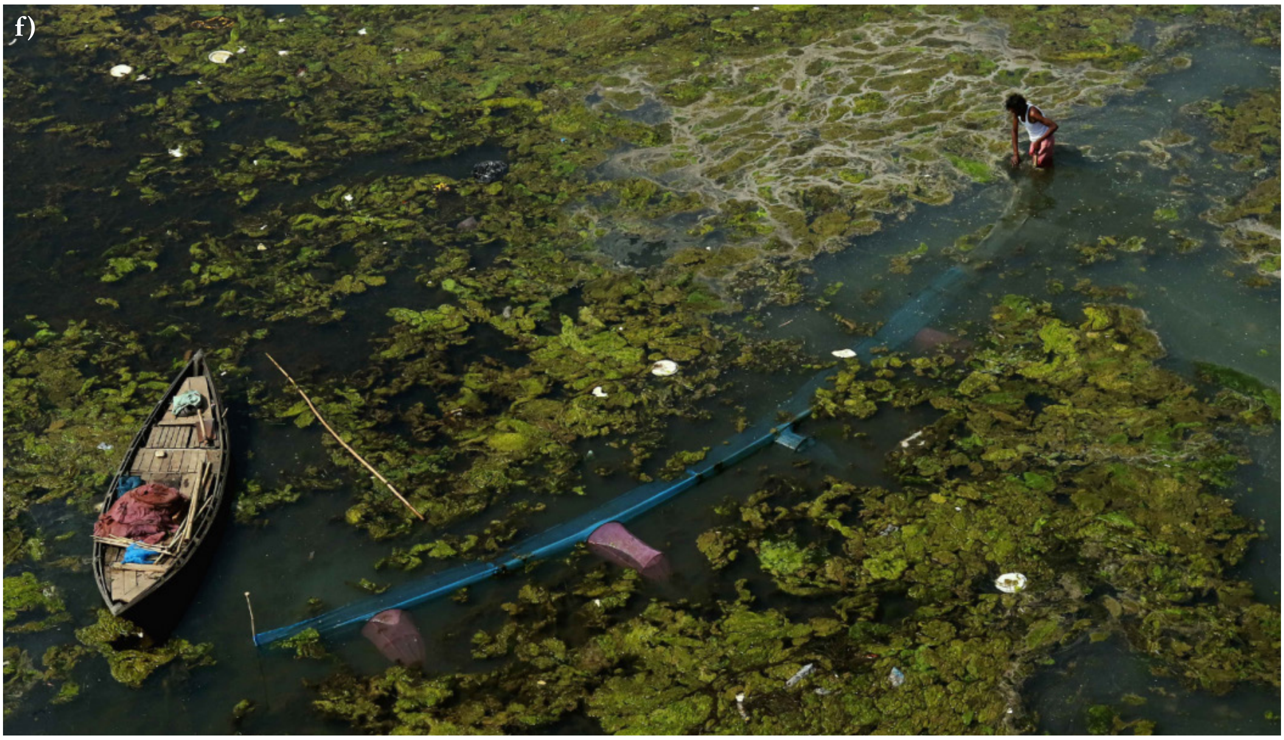

Figure 5.2 Deleterious effects of eutrophication.
a) An algal bloom in Dianchi Lake, China in 2007. Despite millions spent to clean up the lake, the water remains undrinkable and unfit for agricultural or industrial uses. Photo Credit: Greenpeace China.
b) Satellite image of an algal bloom in the Baltic 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).

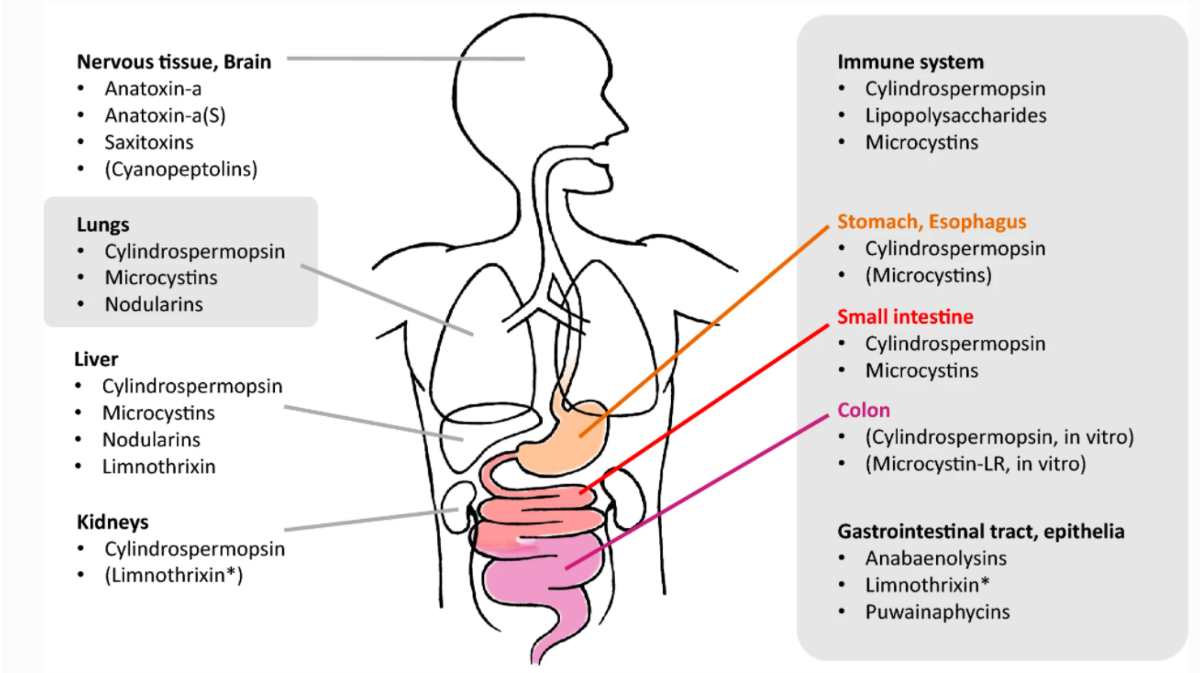

Figure 5.3 Organs/organ systems in humans affected by toxic metabolites of cyanobacteria. Organs traditionally considered targets of toxicity are on the left; organs directly subjected to oral exposure are on the right. Grey-shaded boxes highlight organs with mucosal surfaces that serve as primary entry 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).

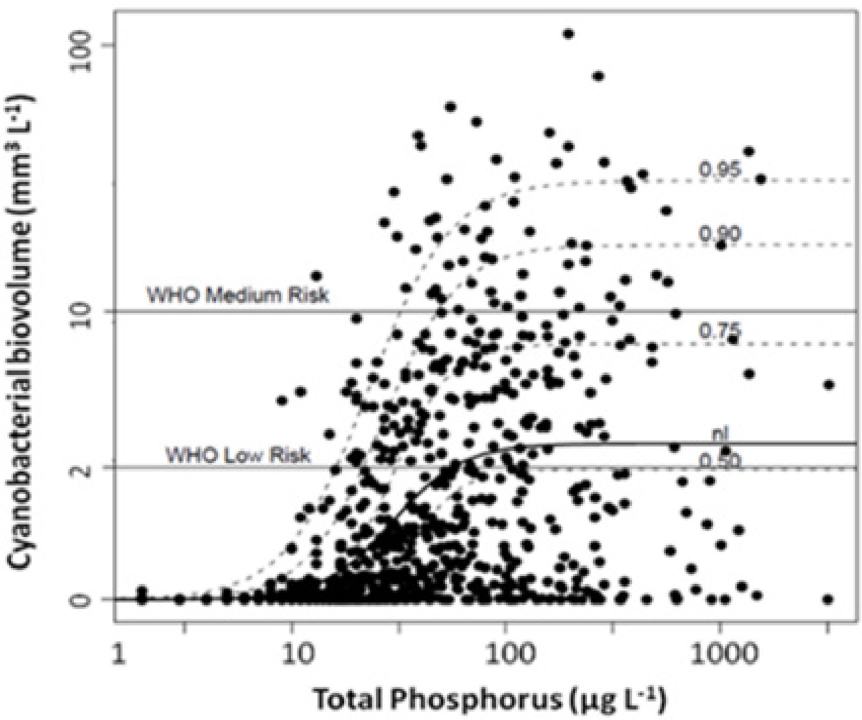

Figure 5.4 Relationship between cyanobacteria biovolume and total phosphorus concentration in > 1500 European lakes in relation to the World Health Organization (WHO) Risk Thresholds for managing human health risk in recreational waters (WHO, 1999). At cyanobacteria biovolumes of between the Low to Medium 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

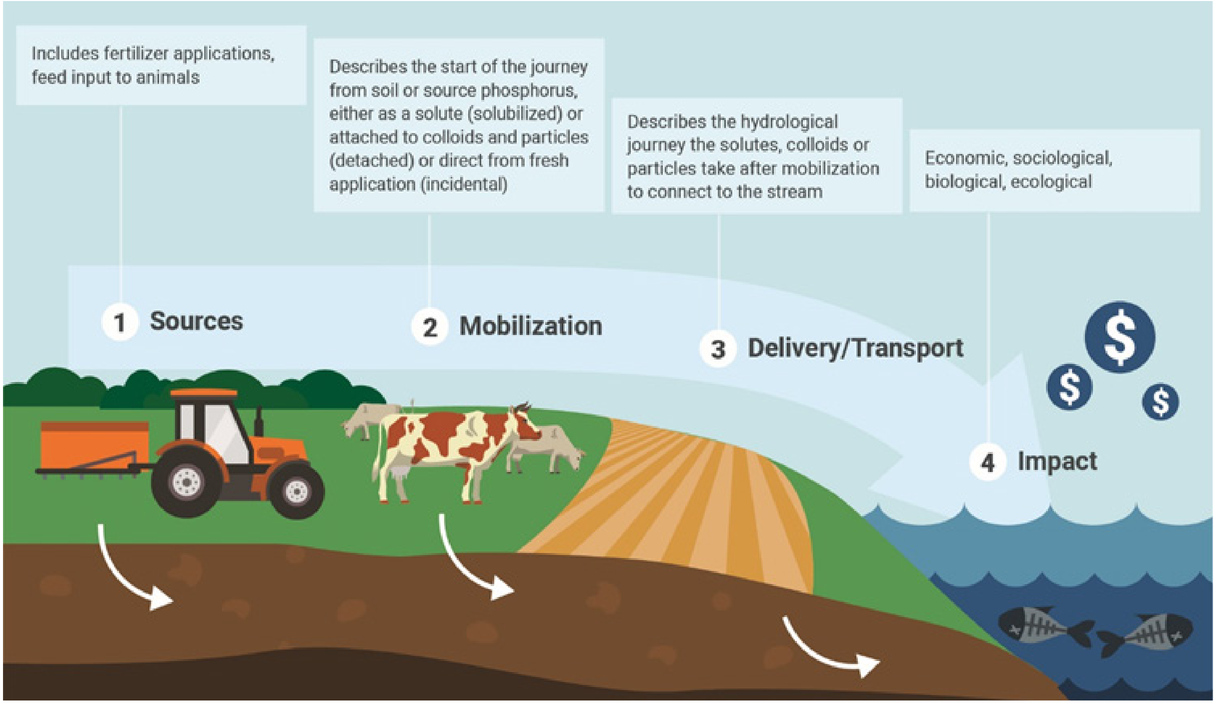

Figure 5.5 The phosphorus (P) transfer continuum. Soil erosion by water has been estimated to contribute to over 50% of total P losses (Alewell et al., 2020), though this can vary markedly between catchments both spatially and temporally (Johnes, 2007b; Lloyd et al., 2019). With respect to the impact 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

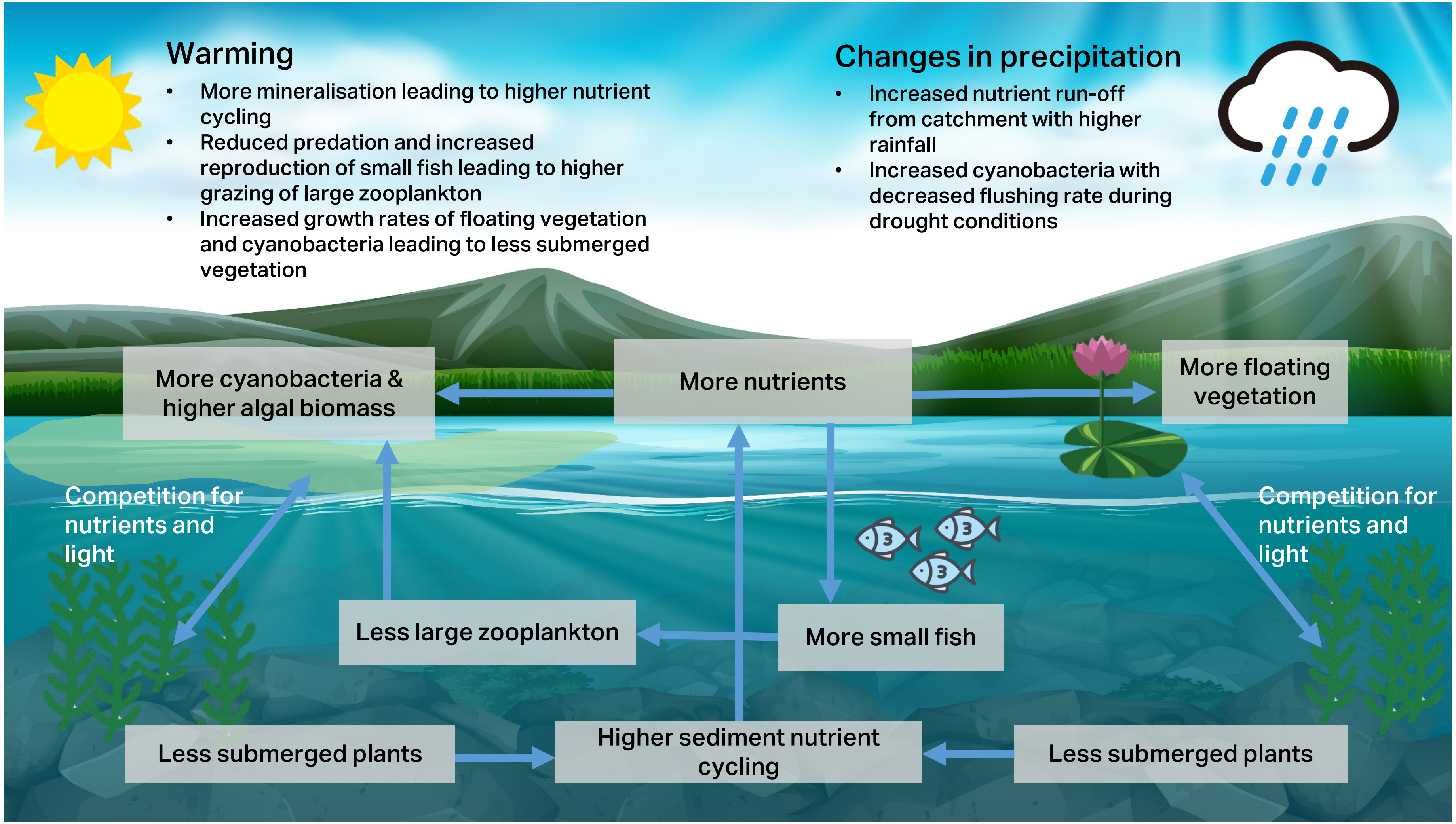

Figure 5.6 Climate change is expected to alter many components of aquatic ecosystems, modifying their responses to nutrient loading. This figure, adapted from Moss et al. (2011), summarise the expected alterations in shallow lake systems, including changes to nutrient loading, concluding that the cumulative 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;

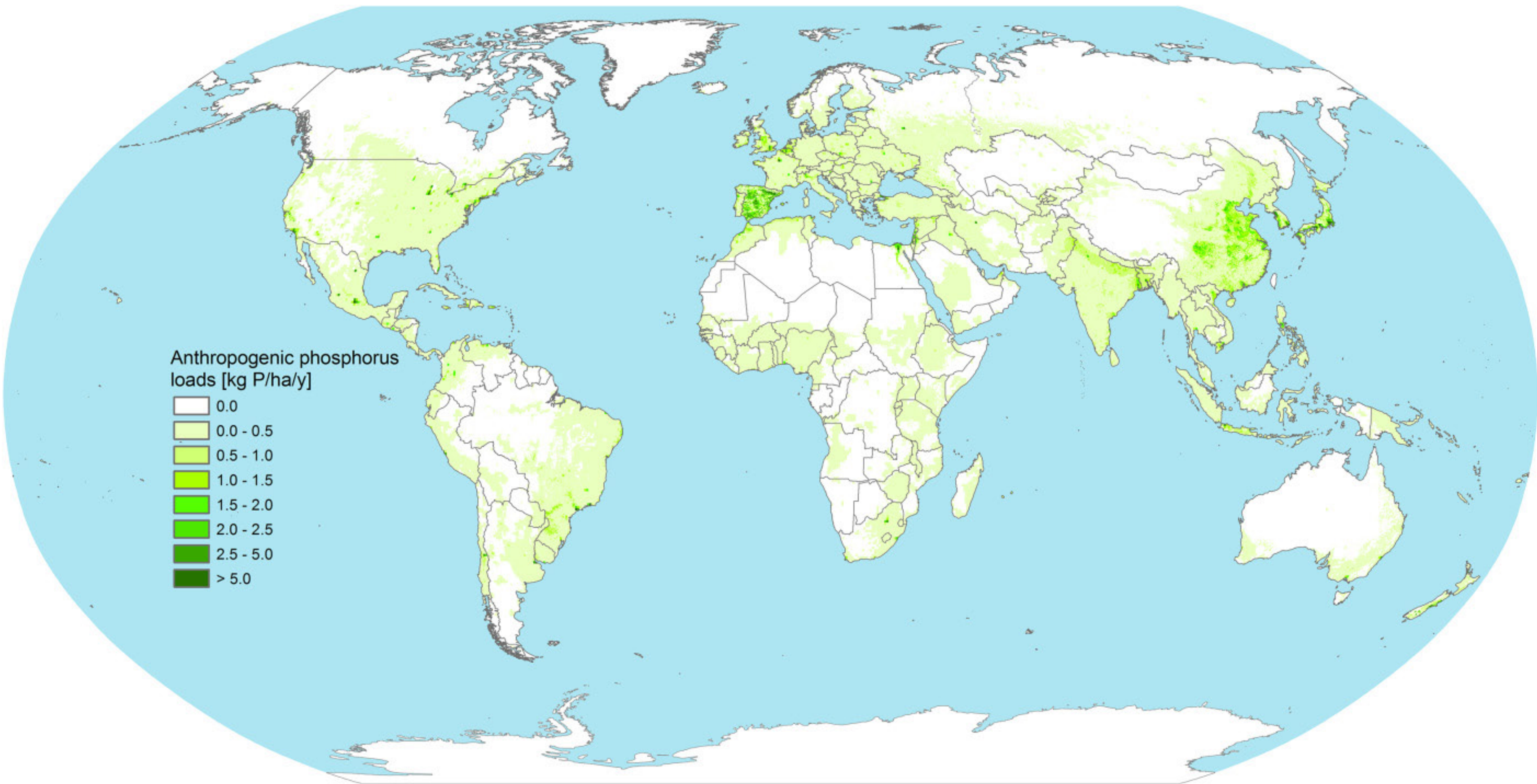

Figure 5.7 Global distribution of anthropogenic phosphorus loads to fresh water from agriculture, industrial, and domestic sectors at a 5 × 5 arc min grid for the period 2002–2010. Image courtesy of Mekonnen and Hoekstra (2018).

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).

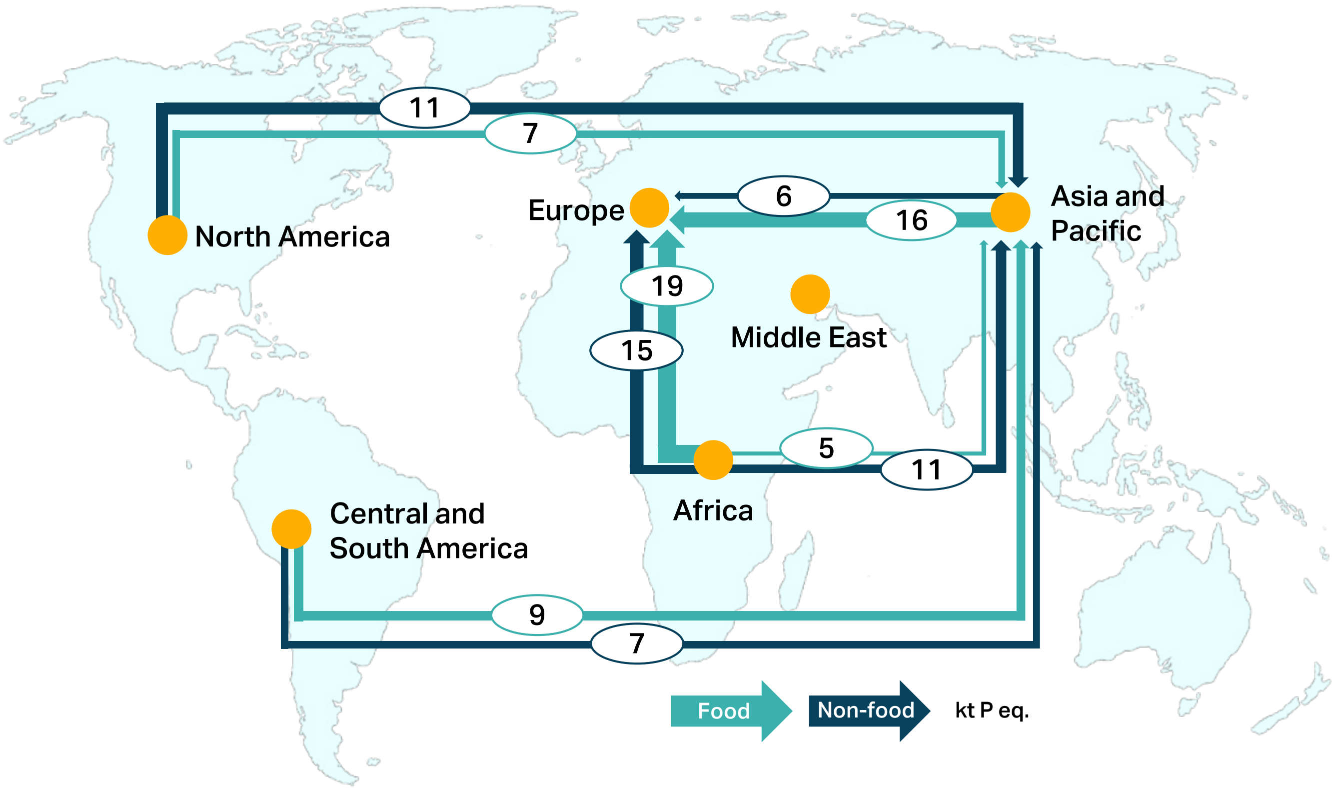

Figure 5.9 International flows of phosphorus (P) in traded food and non-food products, illustrating a complex web of P flows driven by supply and demand drivers globally. The diagram shows the top five continent-level trade-related P displacements of freshwater eutrophication (kt P equivalent year^{-1}). 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.

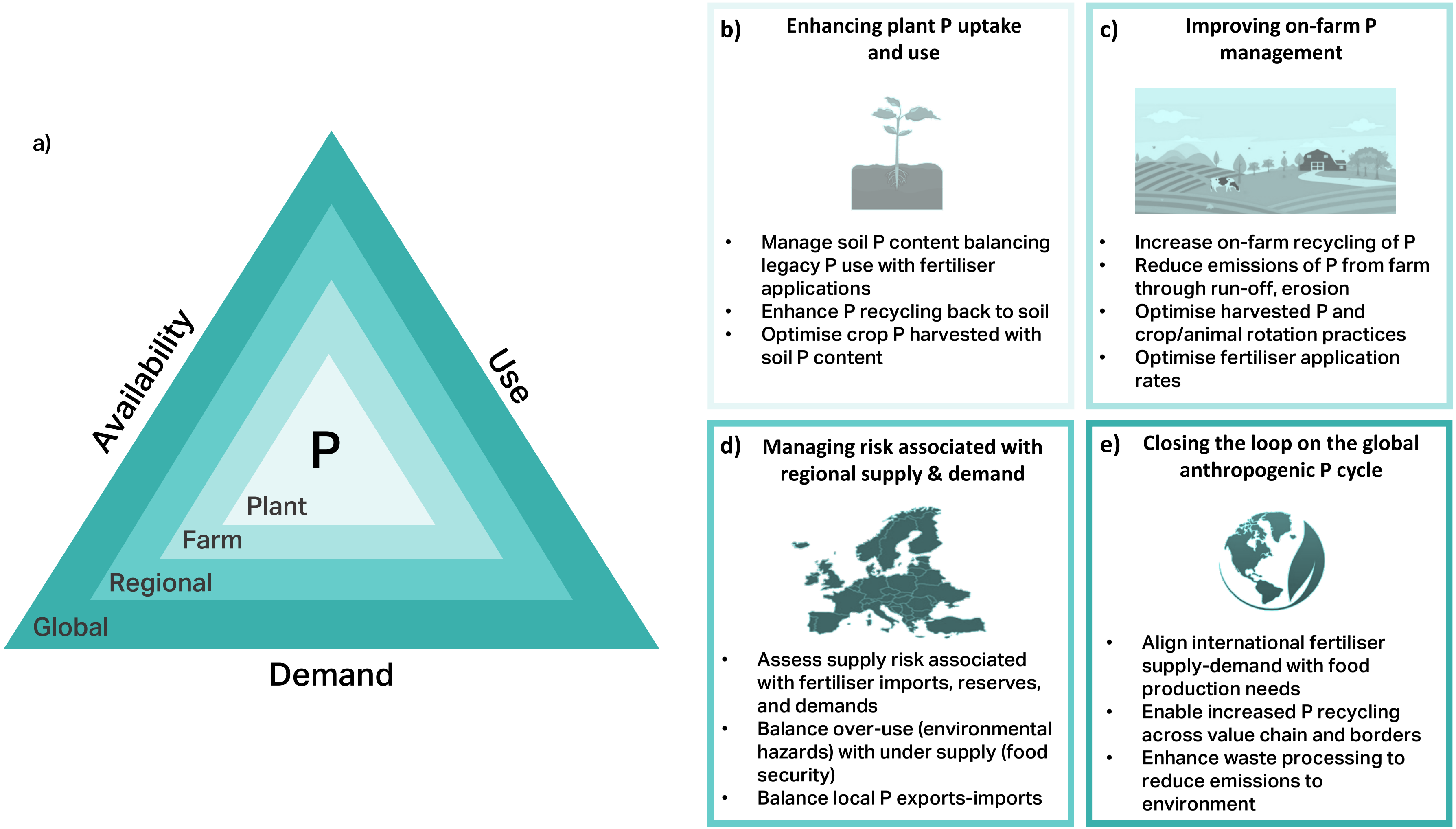

Figure 5.10 Phosphorus (P) conceptual model to optimise P use across scales to ensure (a) equitable and safe P use for all. (b) Plant P dynamics. (c) Optimising on-farm P use. (d) Regional P dynamics for regulation by governing bodies. (e) The ideal global P system of recycling and reducing losses. Image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

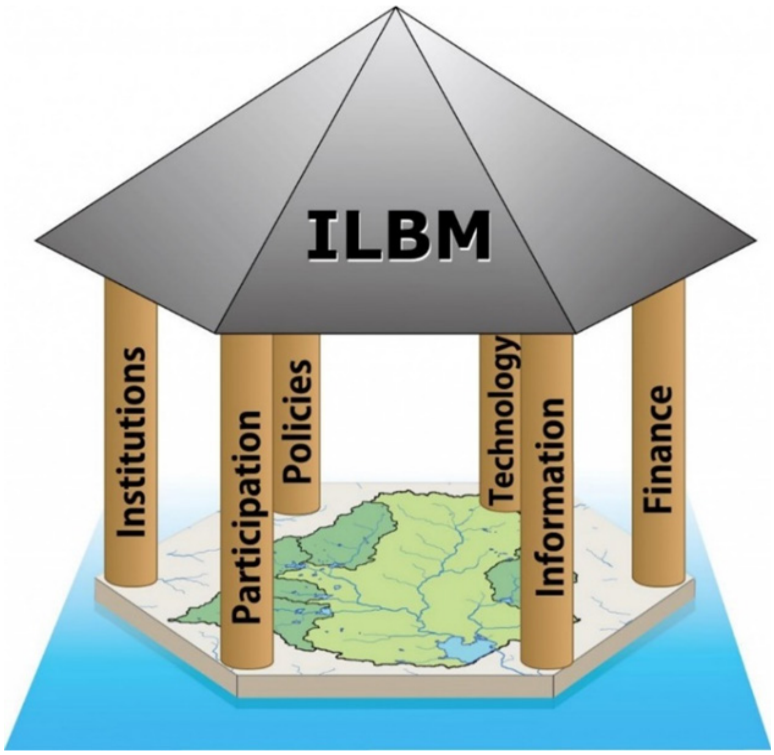

Figure 5.11 The International Lake Environment Committee's Integrated Lake Basin Management (ILBM) framework for sustainable management of lakes and their basins represents a synthesis of experiences from lake management practitioners across the world. This model is scalable to include coupled freshwater 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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