Doctoral Thesis

Improved sustainable phosphorus utilisation by converging market driven functions and requirements with product properties

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Dissertation

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Abstract

Phosphorus is essential for food production, but its current management is inefficient, environmentally harmful and heavily dependent on finite, geopolitically concentrated phosphate rock. At the same time, substantial secondary and legacy phosphorus resources in manure, industrial wastes, wastewater, food-chain residues and soils remain underused, and less than 1% of secondary phosphorus is recycled globally. This work investigates how wastewater-based phosphorus recycling can be implemented as a reliable fertiliser source for crop production in the European Union by assessing its technical, regulatory and economic readiness and its capacity to substitute mineral inputs while maintaining environmental and agronomic safety. First, a global synthesis of secondary and legacy phosphorus resources and a barrier analysis show that manure, mining and fertiliser industry wastes, wastewater and food waste already account for phosphorus flows comparable to current fertiliser demand, while accumulated stocks in soils and sediments exceed it. Yet recycling is constrained by interacting technological, logistical, economic, regulatory and social barriers, highlighting the need for integrated, transdisciplinary approaches that align stakeholder functions and requirements with the properties of recycled products. Second, wastewater-derived fertilisers in Denmark, Germany and Spain are screened against EU and national legislation and agronomic boundary conditions. Only five out of 22 identified technologies fully comply across all three countries, but these compliant routes could meet around 9–22% of wheat, barley and rye phosphorus requirements at national level, with higher contributions in specific regions. The analysis underscores that wastewater recycling is a spatially heterogeneous but non-negligible lever for reducing mineral fertiliser dependence and sewage sludge use. Third, an integrated environmental–economic framework is developed for Spain to compare continued sewage sludge application with struvite, vivianite and calcium phosphate recovered from wastewater. Using data from almost 28,000 agricultural plots, heavy metal accumulation over 50- and 100-year horizons is simulated and combined with a cost–benefit analysis that internalises environmental externalities, including soil remediation and foregone production. The results indicate that continued sludge use can lead to widespread exceedances of zinc, copper and cadmium limits, particularly in alkaline soils, whereas substitution with advanced recycled products substantially reduces these risks. When externalities are included, advanced recycling routes become economically competitive with, and in some cases preferable to, the baseline, even if direct financial returns remain modest. Overall, the present work shows that wastewater-derived fertilisers are not a universal solution, but when

their properties are matched to regulatory, agronomic and economic requirements, they can form a robust component of a diversified, circular phosphorus strategy for European agriculture.

Kurzfassung

Phosphor ist für die Nahrungsmittelproduktion unverzichtbar, doch sein derzeitiges Management ist ineffizient, umweltschädlich und stark abhängig von endlichen, geologisch und geopolitisch konzentrierten Phosphatgesteinsvorkommen. Gleichzeitig bleiben erhebliche sekundäre und im System akkumulierte Phosphorressourcen in Wirtschaftsdüngern, industriellen Abfällen, Abwasser, Reststoffen der Lebensmittelkette und Böden weitgehend ungenutzt, und weniger als 1 % des sekundären Phosphors wird weltweit recycelt. Die vorliegende Arbeit untersucht, wie abwasserbasiertes Phosphorrecycling als verlässliche Düngemittelquelle für den Pflanzenbau in der Europäischen Union implementiert werden kann, indem seine technische, regulatorische und ökonomische Einsatzbereitschaft sowie seine Fähigkeit zur Substitution mineralischer Düngemittelinputs bei gleichzeitiger Wahrung der Umwelt- und agronomischen Sicherheit bewertet werden. Erstens zeigt eine globale Synthese sekundärer und im System akkumulierter Phosphorressourcen in Kombination mit einer Barrierenanalyse, dass Wirtschaftsdünger, Abfälle aus der Bergbau- und Düngemittelindustrie, Abwasser und Lebensmittelabfälle bereits Phosphorströme umfassen, die der heutigen Düngemittelnachfrage vergleichbar sind, während akkumulierte Vorräte in Böden und Sedimenten diese Nachfrage sogar übersteigen. Das Recycling wird jedoch durch miteinander verflochtene technologische, logistische, ökonomische, regulatorische und soziale Hemmnisse eingeschränkt, was die Notwendigkeit integrierter, transdisziplinärer Ansätze verdeutlicht, die Funktionen und Anforderungen der Akteure mit den Eigenschaften recycelter Produkte in Einklang bringen. Zweitens werden aus Abwasser gewonnene Düngemittel in Dänemark, Deutschland und Spanien anhand der EU- und nationalen Rechtsvorschriften sowie agronomischer Randbedingungen geprüft. Nur fünf der 22 identifizierten Technologien erfüllen in allen drei Ländern die gesetzlichen Anforderungen vollständig, doch diese konformen Verfahren könnten rund 9–22 % des Phosphorbedarfs von Weizen, Gerste und Roggen auf nationaler Ebene decken, mit höheren Anteilen in bestimmten Regionen. Die Analyse unterstreicht, dass Abwasserrecycling ein räumlich heterogener, aber keineswegs zu vernachlässigender Hebel zur Verringerung der Abhängigkeit von mineralischen Düngemitteln und der Klärschlammausbringung ist. Drittens wird für Spanien ein integrierter umweltökonomischer Bewertungsrahmen entwickelt, um die fortgesetzte Klärschlammaufbringung mit der Gewinnung und Nutzung von Struvit, Vivianit und Calciumphosphat aus Abwasser zu vergleichen. Unter Verwendung von Daten aus fast 28,000 landwirtschaftlichen Schlägen wird die Schwermetallanreicherung über Zeithorizonte von 50 und

100 Jahren simuliert und mit einer Kosten-Nutzen-Analyse verknüpft, die Umweltexternalitäten, einschließlich Bodensanierung und entgangener Produktion, internalisiert. Die Ergebnisse deuten darauf hin, dass die fortgesetzte Klärschlammanwendung zu weit verbreiteten Überschreitungen der Grenzwerte für Zink, Kupfer und Cadmium führen kann, insbesondere in alkalischen Böden, wohingegen die Substitution durch weiterentwickelte Recyclingprodukte diese Risiken deutlich verringert. Werden Externalitäten einbezogen, so werden die fortgeschrittenen Recyclingverfahren ökonomisch mit dem Referenzszenario konkurrenzfähig und sind diesem in einigen Fällen sogar vorzuziehen, auch wenn die direkten finanziellen Erträge moderat bleiben. Insgesamt zeigt die vorliegende Arbeit, dass abwasserbasierte Düngemittel keine Universallösung darstellen, dass sie jedoch, wenn ihre Eigenschaften auf regulatorische, agronomische und ökonomische Anforderungen abgestimmt werden, einen tragfähigen Bestandteil einer diversifizierten, zirkulären Phosphorstrategie für die europäische Landwirtschaft bilden können.

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1. Introduction

Phosphorus (P) is a fundamental nutrient for plant metabolism and growth. It participates in energy transfer and in the structure of nucleic acids and membranes, and it is required for favourable seed formation, root development and crop maturity (Delgado et al., 2016). Because crop production depends on timely and sufficient P supply, global food security is closely coupled to the availability and efficient use of this nutrient (Cordell et al., 2009). Most anthropogenic P use is directed to fertilisers for food production, which underlines the centrality of P to agricultural output (M. Chen & Graedel, 2016).

Phosphate rock (PR) remains the dominant primary source of P for fertilisers and industrial uses, which anchors agriculture to a finite geological resource with uneven spatial occurrence and variable ore quality (S. K. Haldar, 2018; Reetz, 2016; U.S. Geological Survey, 2024). Concentration of reserves in a few countries and exposure to trade frictions translate into supply risk and price volatility, as illustrated by the sharp fertiliser price spikes of 2008 and subsequent disruptions (Brownlie et al., 2023; Mew, 2016). In Europe, agricultural production depends largely on imported phosphate rock derived inputs, a structural reliance that persists despite decades of policy attention and efficiency gains (van Dijk et al., 2016; Withers et al., 2015). These characteristics of the primary phosphorus supply chain create a persistent vulnerability for food systems and underscore the need to diversify sources and reduce exposure to external shocks.

Linear management of P across extraction, use and disposal has produced persistent environmental and societal burdens. Mismanaged P delivered to freshwaters drives eutrophication, harmful algal blooms, hypoxia and biodiversity loss, with global anthropogenic loads to freshwater on the order of 1.5 Mt P yr^{-1} (Mekonnen & Hoekstra, 2018). Despite policy efforts such as the European Water Framework Directive, many lakes in the region still fail to achieve good ecological status, which signals sustained P pressures and limited restoration outcomes (European Union, 2000). Decades of surplus inputs have also generated legacy P in agricultural soils and lake sediments that continues to fuel diffuse losses, which constrains the effectiveness of downstream mitigation and delays recovery trajectories (Lun et al., 2018; Panagos, Köningner, et al., 2022; Tu et al., 2023). These environmental impacts are matched by sizeable external costs, with recent assessments estimating global costs of P mismanagement in the hundreds of billions of dollars per year, which underscores the societal dimension of a linear P economy and the urgency to shift to circular management (Brownlie et al., 2022).

Secondary sources of P can diversify supply, reduce losses and convert wastes into fertiliser products that support a more resilient agricultural system. Wastewater, manure, food waste and other by-products contain substantial P that today is only partly recovered, which leaves both a resource opportunity and an environmental liability (M. Chen & Graedel, 2016; Cordell et al., 2009). Established and emerging technologies can recover P as mineral fertiliser products including magnesium ammonium phosphate commonly known as struvite, ash derived phosphates from thermal treatment, and precipitated calcium phosphates, as well as processed organic fertilisers from biosolids and digestates (Brownlie et al., 2021; Withers et al., 2015). Among secondary streams, municipal wastewater offers a near term lever in the European context due to existing treatment infrastructure, relatively predictable flows and the possibility to produce standardised products that meet agronomic needs while managing contaminants through process control and product specifications (Withers et al., 2015). Together, these options provide a rationale to complement primary PR with recovered P sources to reduce exposure to supply shocks and to build circularity in food systems (Raniro et al., 2025).

Over the past decade, advances in recycling have expanded the range and maturity of technologies capable of delivering fertiliser-grade P from secondary streams. Full-scale precipitation of struvite at wastewater treatment plants is now well established, motivated by operational benefits and supported by deployment data and process evaluations that document reliable production of standardised granules with low impurity levels (Egle et al., 2015; Jupp et al., 2021). Parallel progress has occurred for ash-derived routes, where sewage sludge ash is processed by wet chemical or thermochemical methods to produce high-purity calcium phosphates with controlled trace metals, and several systems have moved beyond pilot in Europe (Herzel et al., 2016). Vivianite formation under iron-based P removal has opened complementary pathways, either as an iron phosphate product for specific agronomic contexts or as a precursor to calcium phosphate via alkaline post-treatment (Prot et al., 2019). Comparative agronomy indicates that recovered products such as struvite and calcium phosphates can achieve crop responses comparable to conventional references when matched to soil and crop conditions, while offering lower contaminant loads than direct sludge application (Amann et al., 2018; Raniro et al., 2022). These advances provide a technical basis to complement PR with recovered sources at scale, subject to specification and regulatory compliance already mapped in the thesis publications.

Despite a strong body of work on recovery technologies and circular strategies, there is a clear knowledge gap on deployment readiness under European boundary conditions. Existing reviews catalogue options and technical performance, but they seldom test whether products from specific routes can meet current EU and national compliance requirements for agricultural use, and they rarely translate this into realistic regional substitution of mineral inputs (Amann et al., 2018; Herzel et al., 2016; Hukari et al., 2016). Policy analyses describe governance challenges, yet they do not connect regulatory filters and agronomic constraints with quantified supply from wastewater and quantified crop demand at regional scale (Garske et al., 2020; Garske & Ekardt, 2021). Economic studies often omit environmental externalities and system benefits, which limits their relevance for stakeholders who face both compliance obligations and risk exposure (Cordell & Neset, 2014; van Dijk et al., 2016). This thesis addresses that gap by integrating compliance screening for the Fertilising Products Regulation and national rules with agronomic boundary conditions and mass balance scenarios, and by embedding overlooked costs and benefits into an economic framework that speaks to adoption decisions in real settings (European Commission, 2019).

1.1 Aims of the thesis

Because of the previously mentioned gaps, the main objective of this research is to assess how P recycling can be implemented as a reliable source of fertiliser for crop production in the EU, by evaluating its technical, regulatory and economic readiness and its capacity to substitute mineral P inputs while maintaining environmental and agronomic safety.

I. Map the barrier landscape for P recycling across the value chain (Chapter 2).
II. Evaluate wastewater-based technologies according to EU and national compliance and agronomic requirements to identify potential implementation scenarios Chapter 3).
III. Quantify regional substitution of mineral P used in relevant crops by compliant routes under potential P recycling scenarios (Chapter 3).
IV. Develop an environmental–economic framework that internalises externalities and compares recycled products with prevailing practices (Chapter 4).

1.2 Research structure

The present work applies a staged design that links system mapping to deployment analysis and economic evaluation based on three peer-reviewed research articles:

1. The identification of barriers to recycling are organised across technological, regulatory, logistical, market and stakeholder domains using peer reviewed evidence and practitioner sources to identify where adoption stalls across the value chain. The findings were published in the journal *Nature Reviews Earth and Environment* (Raniro et al., 2025) in a shared first authorship with Dr. Henrique Rasera Raniro (see Chapter 2).
2. Wastewater-based routes are screened against current European and national compliance and agronomic requirements, using the EU Fertilising Products Regulation together with the Sewage Sludge Directive and country specific thresholds, to identify technologies that can place products on farms under real boundary conditions in Denmark, Germany and Spain. Compliant P recycling routes are linked to regional substitution by coupling recoverable P from wastewater with crop P demand based on soil pH and available P. The methodology and results of P recycling opportunities were published in the *Journal of Cleaner Production* (Serrano-Gomez et al., 2023) and are presented in Chapter 3.
3. In Chapter 4, an integrated Spanish case study combines a long-term heavy metal accumulation model calibrated with plot data and literature parameters with a cost and benefit framework that internalises environmental externalities and accounts for capital and operating costs of recovery routes, to compare recycled products with prevailing practices and test robustness through sensitivity and scenarios. The results were published in the journal *Waste* (Serrano-Gomez et al., 2025).
4. The general conclusions from the thesis are presented in the last chapter (see Chapter 5), followed by supplementary tables and figures that complement methodologies from Chapter 3 and 4.

2. Overcoming recycling barriers to transform global phosphorus management

2.1 Abstract

The global phosphorus challenge arises from the uneven distribution of phosphorus resources, environmental effects from phosphorus losses and unsustainable linear management. Despite progress in advanced phosphorus recycling, less than 1% of secondary phosphorus resources produced globally are recycled. In this Review, we comprehensively explore global barriers to phosphorus recycling. Manure (15–20 million tons P (MtP) yr^{-1}), mining and fertilizer industry waste (6–12 MtP yr^{-1}), wastewater (~3.7 MtP yr^{-1}) and food waste (~1.2 MtP yr^{-1}) are the major secondary phosphorus resources worldwide. In addition, accumulated legacy phosphorus in soil and sediment comprises a combined stock of more than 3,200 MtP. Phosphorus mismanagement and losses cost stakeholders US$265 billion annually, yet substantial barriers to phosphorus recycling remain. Key challenges to be overcome include low competitiveness of recycled phosphorus products, complex waste handling, limited legacy phosphorus recovery and fragmented collaboration among stakeholders. A shift is needed towards an integrated, systems-based approach that simultaneously addresses technical, economic and societal challenges. Transdisciplinary strategies and research will advance phosphorus recycling and the development of a sustainable, circular phosphorus economy. Incorporating the perspectives of diverse stakeholders will help drive increasingly sustainable phosphorus management.

2.2 Introduction

Phosphorus (P) is crucial for supporting food and industrial production worldwide. Global P demand currently totals 26.5 million tons P (Mt P) yr^{-1}, driven by food production (~80% for fertilizers and 6% for food additives) and industrial applications (14%) (M. Chen & Graedel, 2016; U.S. Geological Survey, 2024). Most consumed P originates from phosphate rock (S. K. Haldar, 2018; Reetz, 2016), a finite resource with known high-quality reserves that are expected to be depleted within the next few centuries (International Fertilizer Industry Association, 2023). In addition, more than 85% of global P deposits are concentrated in just five countries, limiting equitable access to this critical resource, particularly during periods of geopolitical uncertainty (Brownlie et al., 2022; Y. Chen & Chen, 2023). Furthermore, 80–95% of all mined P is lost owing to inefficient P management (Geissler et al., 2018; Scholz & Wellmer, 2015). Much of this loss occurs on farmers' fields, where unconsumed P from past inputs accumulates as legacy P in soil or freshwater sediment. Excess P in aquatic environments causes severe environmental damage. Globally, P mismanagement costs stakeholders approximately US$265 billion annually (Brownlie et al., 2022).

In response to these sustainability, equity and environmental challenges, efforts to establish a circular P economy are gaining momentum (Walsh et al., 2023). For example, China is reducing P waste streams by using industrial sludge as fertilizer (X. Liu et al., 2023). Brazil's National Fertilizer Program is diminishing P inputs in agriculture and reliance on imports by establishing governance and monitoring tools, promoting research and innovation and exploring domestic sedimentary phosphate basins (Ministério da Indústria, 2023; Teodoro et al., 2024). The European Union's Circular Economy Action Plan (Smol, 2023), adopted in 2015, provides an initial framework for nutrient recycling within its territory. Globally, the potential for P recycling is huge, with the amount of P trapped annually in recyclable resources comprising 143% of the current (2024) yearly P demand (M. Chen & Graedel, 2016).

Cascading innovations are focused on the recovery of P from diverse waste streams. The economic, societal and environmental benefits of P recovery (Mayer et al., 2016), the potential of secondary sources of P to partially substitute mineral P in the production of fertilizers (Devault et al., 2025; Egle et al., 2016) and a role for recycling in closing nutrient loops (Jupp et al., 2021) highlight the importance and advantages of P recycling. Concurrently, technological breakthroughs have facilitated the transition of large-scale operations to the use of secondary P in manufacturing industrial products (Devault et al., 2025; Egle et al., 2016). Despite notable progress in infrastructure (Bernal et al., 2015; Commission, 2021), policy development (Garske & Ekardt, 2021; Kalpakchiev et al., 2023) and recycling technologies (Egle et al., 2016; Jupp et al., 2021) for circular nutrient management, challenges in P management persist. Contextual differences, disciplinary fragmentation and limited stakeholder coordination hinder the development of well-defined pathways to achieve greater P circularity worldwide (Walsh et al., 2023; F. Zhu et al., 2023), reflecting its nature as a wicked problem (a complex societal issue that is difficult to solve owing to its interconnectedness, lack of definition and absence of a definitive solution).

In this Review, we explore the reasons why P recycling from secondary sources remains limited. We discuss a range of technical, economic and societal barriers to global P recycling (Aarikka-Stenroos et al., 2023; Cordell & White, 2014; Hosseinian et al., 2023) and argue that fostering transdisciplinary collaborations is essential to improving P sustainability worldwide, precisely because such collaborations are best positioned to align the diverse interests of stakeholders across sectors and disciplines.

2.3 The need to improve phosphorus management

The ideal P value chain is circular, but in practice it is predominantly linear. The cycle begins when P mined from phosphate rock is used to produce fertilizers and other goods. After manufacturing, P is consumed by humans, crops or animals, but substantial amounts of P are lost to secondary P waste streams at every stage of the P life cycle. P losses to the environment are also pervasive, leading to P accumulation in soil and sediment, where it can contribute to environmental degradation (Geissler et al., 2018; Lou et al., 2018; Pavinato et al., 2020; Solangi et al., 2023).

Furthermore, as access to phosphate rock reserves diminishes, both economically and geographically, the risks to global food production rise (B. Li et al., 2023; Powers et al., 2019). These challenges highlight the need to develop strategies for recovering and recycling P. The European Union (EU) has some of the most advanced technology, data availability and legislation relating to P recycling yet, even in the EU, the P cycle remains essentially linear (Figure 1).

2.3.1 Geopolitical dependence on primary phosphorus imports

Most regions worldwide rely heavily on P imports from a limited number of countries (Morocco alone contains ~67% of the world's known reserves (Cordell & Neset, 2014; Nanda et al., 2019; U.S. Geological Survey, 2024)), creating pronounced geopolitical vulnerabilities in food security (Bonini & Wesenbeeck, 2023; Pistilli, 2025). Political, economic and environmental disruptions can lead to global price shocks (Bonini & Wesenbeeck, 2023). For example, in 2008, phosphate fertilizer prices spiked by 800%, mostly due to rising energy costs and geopolitical trade policies (Mew, 2016), generating supply constraints and threatening agricultural production and food security in several parts of the world (Cordell et al., 2009; Cordell & White, 2011). More recent supply chain disruptions include the COVID-19 pandemic (Moosavi et al., 2022), the Ukraine–Russia conflict (Kee et al., 2023), reductions in P fertilizer exports by China and Russia (Chow & Patton, 2022; Pistilli, 2025), and trade wars between key stakeholders (such as China and the USA (White, 2025)). The negative effects of these events disproportionately affected countries in Africa (Hebebrand & Glauber, 2024). Asymmetric risks associated with primary P dependence underscore the need for diversified P sources and more resilient supply chains.

2.3.2 Agricultural needs and phosphorus wastage

Approximately 90% of all mined P resources are used in fertilizers (Brunner, 2010; Ridder et al., 2012), and global P use since the Green Revolution has increased more than sixfold (Brownlie et al., 2023; M. Chen & Graedel, 2016). However, quantifying global P flows and stocks is challenging owing to major regional differences in P resource availability, prices and efficiency of use, all of which are shaped by economic, political and environmental factors (Brownlie et al., 2023; Walsh et al., 2023). The most comprehensive accountings of European and global P mass describe conditions in 2005 (van Dijk et al., 2016) and 2009 (Cordell et al., 2009), respectively. Both assessments highlight increasing agricultural demand and pervasive losses throughout the P life cycle.

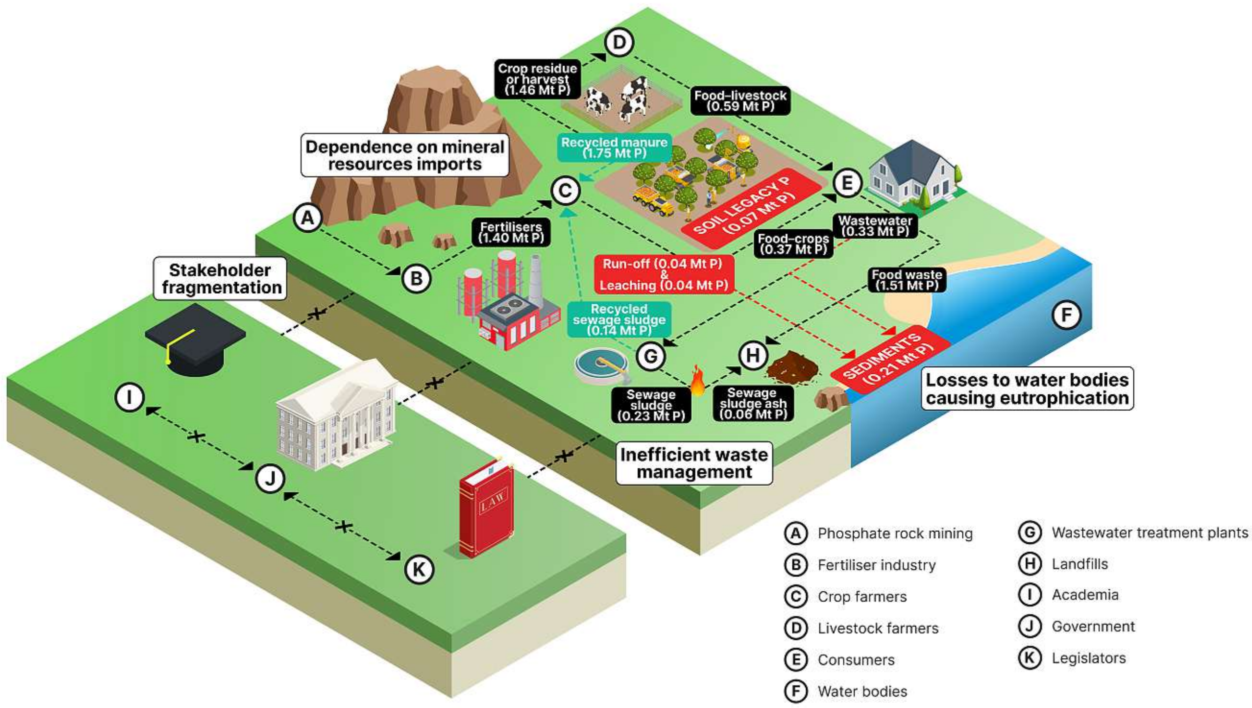

Figure 1. Phosphorus cycle flows and challenges in the EU. Phosphorus flows in the European Union (EU) in 2005, based on data from (van Dijk et al., 2016). The predominantly linear cycle begins with phosphate rock mining (mostly outside the EU), followed by phosphorus use in fertilizers, human and animal consumption, and eventual inefficient disposal as waste, leading to widespread losses to the environment as legacy phosphorus accumulates in soil and sediment. Partial recycling of phosphorus occurs through manure and sludge application in agriculture, but addressing environmental and health effects remains technically and logistically challenging. No major advanced recycling efforts had been established when these data were collected. Collaboration-hindering stakeholder fragmentation is a major barrier to large-scale implementation of advanced phosphorus recycling initiatives.

Where intensive agricultural practices coincide with widespread access to mined P resources, high P input and legacy P often result (Lun et al., 2018). China and Brazil use substantial amounts of mineral fertilizer (more than 32 kg P ha^{-1} of cultivated land per year) (Lun et al., 2018) as P input compared with 5–10 kg P ha^{-1} yr^{-1} of manure, and P removal through crop harvests is also high (24 and 26 kg P ha^{-1} yr^{-1}, respectively). Conversely, the low agricultural productivity in most African countries is partially attributed to low P input (Vanlauwe et al., 2023) (Figure 2).

Most European countries have moderate or low mineral fertilizer inputs (0–16 kg P ha^{-1} yr^{-1}) and moderate or high manure inputs (8–24 kg P ha^{-1} yr^{-1}), except for the Netherlands and Belgium, where manure application is remarkably high (65 and 35 kg P ha^{-1} yr^{-1}, respectively). In the USA, mineral fertilizer application is moderate (8–16 kg P ha^{-1} yr^{-1}), manure input is low (0–8 kg P ha^{-1} yr^{-1}) and crop removal is moderate (8–16 kg P ha^{-1} yr^{-1}) (Lun et al., 2018). Importantly, P losses are extensive almost everywhere. For example, the terrestrial P surplus (that is, P that was applied to fields but not exported by crop harvest) in the USA in 2012 is estimated at 1.85 Mt P, with many agricultural areas exhibiting high surpluses, particularly in the upper Midwest (Sabo et al., 2021).

2.3.3 Environmental effect of phosphorus mismanagement

P mismanagement results in massive environmental degradation. Some mismanaged P ends up in water bodies and causes eutrophication (W. Liu et al., 2020; Wang et al., 2022), resulting in harmful algal blooms, dead zones and biodiversity loss. Annually, approximately 1.5 Mt of anthropogenic P are lost to freshwater systems (Mekonnen & Hoekstra, 2018). The EU released a Water Framework Directive more than 25 years ago (European Union, 2000), yet 60% of lakes in the region do not meet the directive's 'good' standard. Pervasive eutrophication and ineffective water restoration highlight the urgent need to rethink P use practices and develop sustainable P management solutions.

By recovering P from nutrient-rich waste streams, such as wastewater and animal manure, using methods such as chemical precipitation, advanced composting and biochar production, P recycling from agricultural systems can be increased (M. Chen & Graedel, 2016; Egle et al., 2016; Powers et al., 2019). Furthermore, recovering recalcitrant P stocks in soil and sediment can be crucial to ensuring global P circularity.

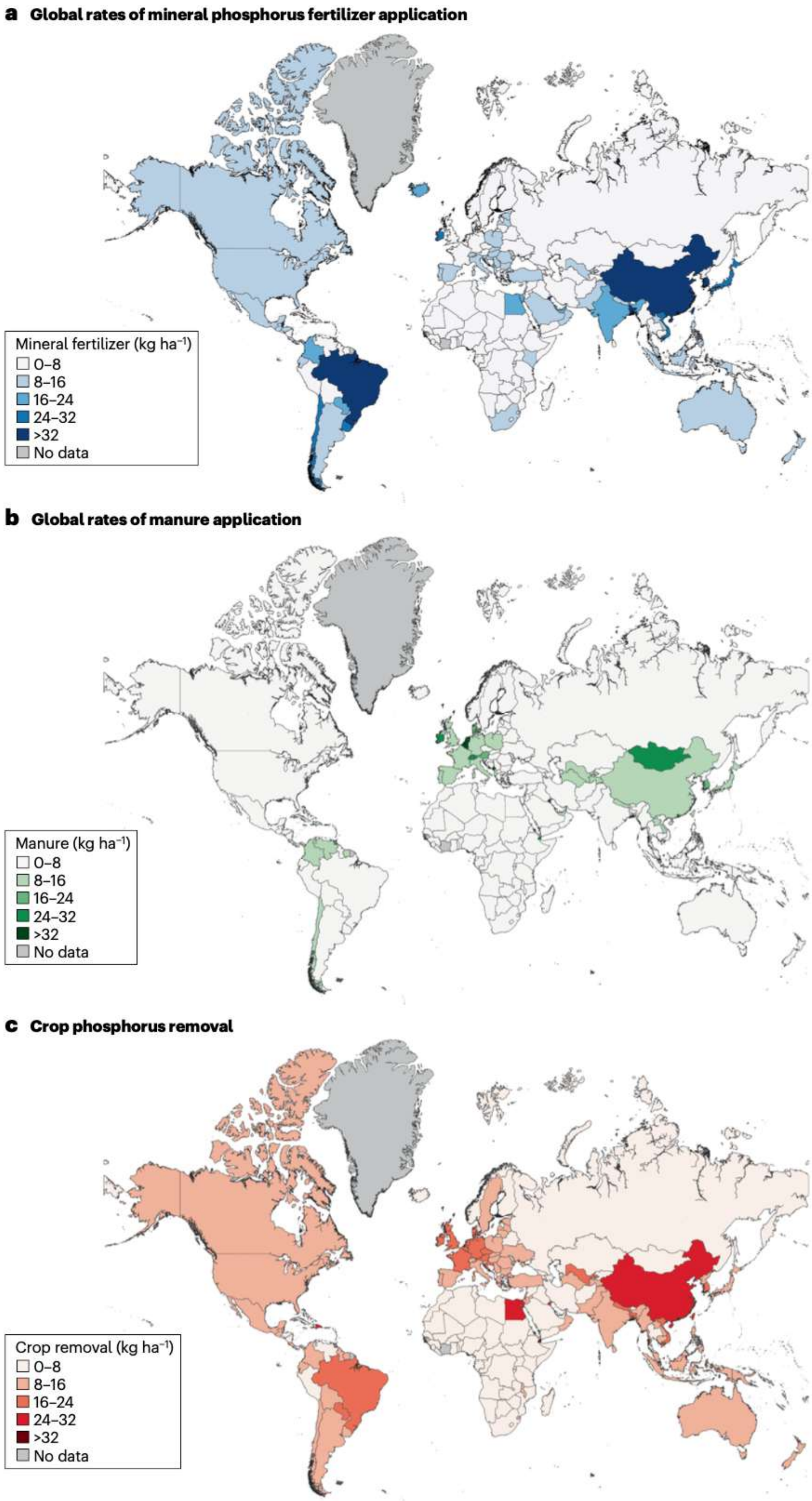
a Global rates of mineral phosphorus fertilizer application
Mineral fertilizer (kg ha^{-1})
0–8
8–16
16–24
24–32
>32
No data
b Global rates of manure application
Manure (kg ha^{-1})
0–8
8–16
16–24
24–32
>32
No data
c Crop phosphorus removal
Crop removal (kg ha^{-1})
0–8
8–16
16–24
24–32
>32
No data

Figure 2. The major inputs of phosphorus in agriculture are through mineral fertilizer and manure application, whereas crop offtake is the main process of phosphorus removal. a, Global rates of mineral phosphorus fertilizer application. b, Global rates of manure application. c, Crop phosphorus removal. The maps highlight the uneven patterns of phosphorus consumption around the world, with disproportionally high phosphorus inputs in countries such as Brazil and China, higher manure application in Europe, China and Mongolia, and higher phosphorus uptakes in Europe despite low mineral phosphorus inputs, which is attributed mainly to legacy phosphorus draw-down. Based on data from (Lun et al., 2018).

These geopolitical, wastage and environmental aspects reveal a global P landscape marked by stark regional contrasts, ranging from overapplication and environmental losses to chronic underuse owing to limited access to P resources. Addressing these imbalances is essential to improving agricultural productivity and the sustainability of P use worldwide.

2.4 Recovery of secondary phosphorus

Improving P circularity is indispensable to overcoming the geopolitical, agricultural and environmental challenges associated with P mismanagement (Geissler et al., 2018; Walsh et al., 2023). Achieving this goal requires optimizing the use of primary P resources (mined phosphate rock), using P stocks accumulated in soil (Lou et al., 2018; Solangi et al., 2023) and sediment (Tu et al., 2023), and enhancing P recycling from P-rich secondary sources, including waste from the mining and fertilizer industries (Bilal et al., 2023; M. Chen & Graedel, 2016), livestock manure (Devault et al., 2025), wastewater and sewage sludge (Carrillo et al., 2024; Jama-Rodzeńska et al., 2021), and food waste (Girotto et al., 2015) (Table 1).

Table 1. Major secondary phosphorus resources, recycling techniques and challenges

Secondary p waste stream or legacy P stock	Global amount generated	Current fate	Potential advanced recycling techniques for secondary P or draw-down strategies for legacy P	Challenges
Secondary phosphorus resources				
Mining and fertilizer industry	6–12 $MtPyr^{-1}$ (M. Chen & Graedel, 2016; Fuleihan, 2012; Geissler et al., 2018;	Disposal in landfills or coastal waters, or stacked on land	Pyrolysis, leaching and precipitation	High complexity and cost

	U.S. Geological Survey, 2024)			
Livestock manure	15–20 MtPyr^{-1} (M. Chen & Graedel, 2016)	Direct application (~90%)	Flocculation, settling, screw pressing, belt filtration, centrifugation and anaerobic digestion (and subsequent dissolved air flotation)	Large volumes produced only in a few regions Geographic disconnect between generation and use locations Dewatering is energy-intensive and transportation is logistically complex
Food waste	~1.2 MtPyr^{-1} (M. Chen & Graedel, 2016; Girotto et al., 2015)	Landfill (~60%) Animal feed (5-10%)	Composting, anaerobic digestion and incineration	Low phosphorus concentration High heterogeneity of materials
Wastewater	~3.7 MtPyr^{-1} (Jones et al., 2021; Kok et al., 2018)	Discharge into surface waters, irrigation and agricultural reuse[b]	Biological removal and chemical precipitation	Requires infrastructure Processes might substantially differ depending on waste characteristics and regions
Sewage sludge[a]	~3 MtPyr^{-1} (Witek-Krowiak et al., 2022)	Landfill, direct application[b]	Incineration, pyrolysis and composting	Yuck factor Low phosphorus availability in sludge-derived products Chemically intensive
Sewage sludge ash[a]	0.13–0.22 MtPyr^{-1} (L. Fang et al., 2021)	Landfill, direct application[b]	Chemical and thermochemical treatments	Energy and chemically intensive High costs
		Legacy phosphorus stocks		
In soil	~815 MtP (Sattari et al., 2012)	Accumulation, erosion and run-off	Phosphorus-mining crops, biostimulants, bioengineering, plant breeding and biofertilizers	Low bioavailability Geographical dispersion
In sediment	>2,600 MtP (Tu et al., 2023)	Accumulation	Direct application	Dredging lakes is costly and complex Contamination is possible

[a]The phosphorus reported in sewage sludge and sludge ash originates from the total phosphorus in wastewater. These values are not additive; instead, the phosphorus content in sludge and ash is contained within the original wastewater phosphorus flow. [b]Subject to local legislation. MtP, million tons phosphorus.

2.4.1 Mining and fertilizer industry waste

The mining industry produces substantial amounts of secondary P contained in by-products and waste from rock mining (1.1–3.0 Mt yr^{-1}) and phosphoric tailings generated during rock beneficiation (2.3–4.6 Mt yr^{-1}), which together represent 12–16% of total P mined globally (Scholz & Wellmer, 2015; U.S. Geological Survey, 2024). Although P recovery from mining waste using pyrolysis, leaching and precipitation have been proposed, these approaches are complex and expensive (Xiao et al., 2024; Yu et al., 2024), and large-scale recovery is currently not viable

(Spooren et al., 2020). Typically, mining waste is landfilled, and usually covered with vegetation to reduce environmental risks such as erosion, dust production and P run-off.

The fertilizer industry also generates P-rich waste such as P slag (~1.8 MtP yr^{-1}) and ferrophosphorus (~0.3 MtP yr^{-1}) (M. Chen & Graedel, 2016). In addition, ~300 Mt of phosphogypsum (containing 6–9.8 MtP) is produced annually by acidifying phosphate rock (M. Chen & Graedel, 2016; Singh, 2002). Although phosphogypsum has agronomic applications, 58% is landfilled (Fuleihan, 2012) and 28% is discharged into the sea (Pliaka & Gaidajis, 2022), comprising a global stockpile of 60–160 Mt of P (Bilal et al., 2023; Pliaka & Gaidajis, 2022), of which only 14% is further treated (Bilal et al., 2023).

2.4.2 Livestock manure

Livestock manure is the largest secondary P resource (15–20 MtP yr^{-1}) (M. Chen & Graedel, 2016; van Dijk et al., 2016), accounting for more than 50% of the annual secondary P generated worldwide. In the EU, ~2.2 MtP yr^{-1} comes from manure, of which ~90% is directly applied to agricultural land (Devault et al., 2025; Köninger et al., 2021; Panagos, Köningner, et al., 2022). However, a substantial portion of manure is difficult to recover because grazing animals deposit it directly onto grasslands (Köninger et al., 2021). Intentional application is often concentrated near livestock production areas, serving more as a means of waste disposal than as a targeted agronomic strategy (Devault et al., 2025). This practice can lead to the accumulation of soil legacy P. Moreover, manure typically has a low N:P ratio, which can contribute to P overapplication as farmers prioritize addressing crop nitrogen requirements. In China, ~2.14 MtP yr^{-1} from manure is applied on agricultural land (Q. Zhang et al., 2023), representing 26% of the country's total P demand, but this is only ~50% of the P that could be harnessed from manure domestically (Bai et al., 2016).

Owing to the low P concentration in manure (<1% of P per fresh weight), concentrating P from this source can improve P recovery and subsequent recycling, thereby facilitating efficient handling, transportation and application (Devault et al., 2025; Köninger et al., 2021). The water content and volume of manure can be reduced using non-thermal methods, such as flocculation, settling, screw pressing, belt filtration centrifugation and dissolved air flotation after anaerobic digestion (Hjorth et al., 2010; Hjorth & Jørgensen, 2012), but technical, logistical and financial challenges remain.

Biogas production from manure is common in developed countries, yielding a digestate containing ~2% P (Möller & Müller, 2012). To further concentrate P in fresh manure and its digestate, solid–liquid separation techniques have proved effective (Kabeyi & Olanrewaju, 2022). About 70–75% of P can be recovered in the solid fraction without flocculation, whereas flocculation increases the recovery rate to 80–90%. Owing to the low P concentration in manure (<1% of P per fresh weight), concentrating P from this source can improve P recovery and subsequent recycling, thereby facilitating efficient handling, transportation and application (Devault et al., 2025; Köninger et al., 2021). The water content and volume of manure can be reduced using non-thermal methods, such as flocculation, settling, screw pressing, belt filtration centrifugation and dissolved air flotation after anaerobic digestion (Hjorth et al., 2010; Hjorth & Jørgensen, 2012), but technical, logistical and financial challenges remain.

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2.4.3 Food and biorefining wastes

As much as 1.2 MtP could be recycled annually from farming, food manufacturing, biorefining, and consumer wastes and by-products (M. Chen & Graedel, 2016). However, resource heterogeneity complicates recycling from food waste (Hosseinian et al., 2023). Recovery techniques such as composting, anaerobic digestion and fermentation are the most common strategies for nutrient recovery from food waste. Other methods, such as incineration, are promising, but come with their own drawbacks, such as increased pollution (Girotto et al., 2015; D. Haldar et al., 2022).

Industrial food manufacturing is a major source of food waste, accounting for almost 50% of the food waste in the entire supply chain in the UK (Garcia-Garcia et al., 2017). This waste often takes the form of useful materials such as whey and starch residues (Garcia-Garcia et al., 2017). Meat and bone meal (a slaughterhouse by-product) has high P content (3–5%) and are already used as a P fertilizer in some countries, but concerns about pathogen risk and public perception still limit its use in other countries (Otles et al., 2015).

Biorefinery residuals, such as waste from bioethanol production or breweries, are also rich in P. A prominent example is distillers' dried grains with solubles, a by-product of corn ethanol production, of which ~38 Mt yr^{-1} is produced in the USA alone. With a typical P concentration of ~1% of dry weight (D. Haldar et al., 2022; Otles et al., 2015), this amounts to ~0.38 MtP yr^{-1}, making it one of the largest flows of concentrated organic secondary P in North America (Ruffatto et al., 2023). Although commonly used in livestock feed, which aids P recycling to some extent, the spatial disconnect between production and consumption sites requires complex transport logistics and high costs, often resulting in ineffective P recovery (Ruffatto et al., 2023). Biorefinery residuals, such as waste from bioethanol production or breweries, are also rich in P. A prominent example is distillers' dried grains with solubles, a by-product of corn ethanol production, of which ~38 Mt yr^{-1} is produced in the USA alone. With a typical P concentration of ~1% of dry weight (D. Haldar et al., 2022; Otles et al., 2015), this amounts to ~0.38 MtP yr^{-1}, making it one of the largest flows of concentrated organic secondary P in North America (Ruffatto et al., 2023). Although commonly used in livestock feed, which aids P recycling to some extent, the spatial disconnect between production and consumption sites requires complex transport logistics and high costs, often resulting in ineffective P recovery (Ruffatto et al., 2023).

Post-consumer food waste, primarily from households and the food service industry, is the largest global food waste stream, amounting to ~570 Mt annually (Chia et al., 2024). This waste is typically heterogeneous and often contaminated with plastics or packaging residues (Schanes et al., 2018), posing major logistical and regulatory challenges for safe reuse (Arcas-Pilz et al., 2023). In Barcelona, compost derived from household food waste is applied to urban agriculture plots, but contamination and legal hurdles intended to lower the risk of contamination limit broader nutrient recycling efforts (Arcas-Pilz et al., 2023). Another example is seen in Thailand, where food waste composting and direct use as animal feed recovered up to 71% of the P content in food waste from retail and wholesale markets (Mokjatturas et al., 2025). Despite its potential, P recovery from food waste remains underdeveloped, partially owing to severe logistical, environmental and behavioural challenges, low economic incentives and weak regulatory enforcement (D. Haldar et al., 2022; Otles et al., 2015).

2.4.4 Wastewater

Wastewater refers to water discarded from households, businesses and industries, as well as stormwater run-off. Globally, ~360 billion m^3 of wastewater is produced annually, containing

3.7 MtP (Jones et al., 2021; Kok et al., 2018). However, in current market conditions, the economic feasibility of P recycling from wastewater remains limited (Kok et al., 2018), so most wastewater treatment plants prioritize meeting discharge requirements for treated wastewater rather than actively recovering P (F. Zhu et al., 2023). P is typically removed from wastewater through chemical, biological (Diaz et al., 2022) or combined treatment methods (Korving et al., 2018), with ~90% ending up in sewage sludge (Witek-Krowiak et al., 2022).

Sewage sludge is sometimes applied directly to agricultural fields (Danish Ministry of the Environment, 2013). However, this practice is being increasingly restricted owing to concerns about contaminants, including heavy metals, pathogens, microplastics and toxic organic compounds. Although only a small fraction of P from wastewater is recovered using advanced recovery methods (ESPP, 2023), the potential of numerous technologies has been demonstrated at full or pilot scale (Egle et al., 2015; ESPP, 2023), particularly in North America, Europe and Asia. For effective recovery, P must first be dissolved and then concentrated and recovered through precipitation, crystallization or adsorption (Egle et al., 2015). Typically, sludge liquor, sludge or sludge ash are used to produce P-recovered products such as struvite, calcium phosphate or phosphoric acid (Wijdeveld et al., 2022).

Recovering P directly from sludge tends to yield lower-purity struvite (with lower P and higher impurity concentrations), whereas recovery from the water separated from sludge generally produces a higher-quality product and higher yield but, combined, these methods only capture 10–40% of influent P (Egle et al., 2016). High recovery yields, ranging from 60% to 70%, can be achieved through vivianite precipitation (Wijdeveld et al., 2022) or sludge acidification, followed by solid–liquid separation and precipitation (Quist-Jensen et al., 2018). However, these processes are energy intensive and thus contribute to global warming, and they require substantial amounts of acid (Amann et al., 2018). Although the P recovery efficiency is higher than for struvite-based technologies, these methods have yet to progress beyond pilot-scale demonstrations (Wijdeveld et al., 2022).

High yields of micropollutant-free P, typically 80–90%, can also be recovered from sludge ash (Egle et al., 2015). Current technologies for P extraction from sludge ash can be categorized as wet-chemical, thermochemical and electrodialysis methods (Y. Zhu et al., 2022). These methods vary in their effectiveness in heavy metal removal, emissions production and energy demands (Amann et al., 2018). P extraction from sludge ash can recover high volumes of P, making centralized facilities feasible. Other methods, such as pyrolysis or hydrothermal carbonization, produce char that can be used directly on agricultural fields, if inorganic and organic pollutants are below local legal thresholds (Y. Zhu et al., 2022). Despite the high potential for P recovery from wastewater and its by-products, only 61 facilities worldwide currently pursue advanced P recovery that goes beyond conventional sludge recycling or land application of biosolids, collectively recovering ~$4.2 \times 10{-}3$ MtP yr^{-1} from secondary sources [88] (Figure 3). High capital expenditure requirements hinder investment in these promising technologies (Egle et al., 2015, 2016).

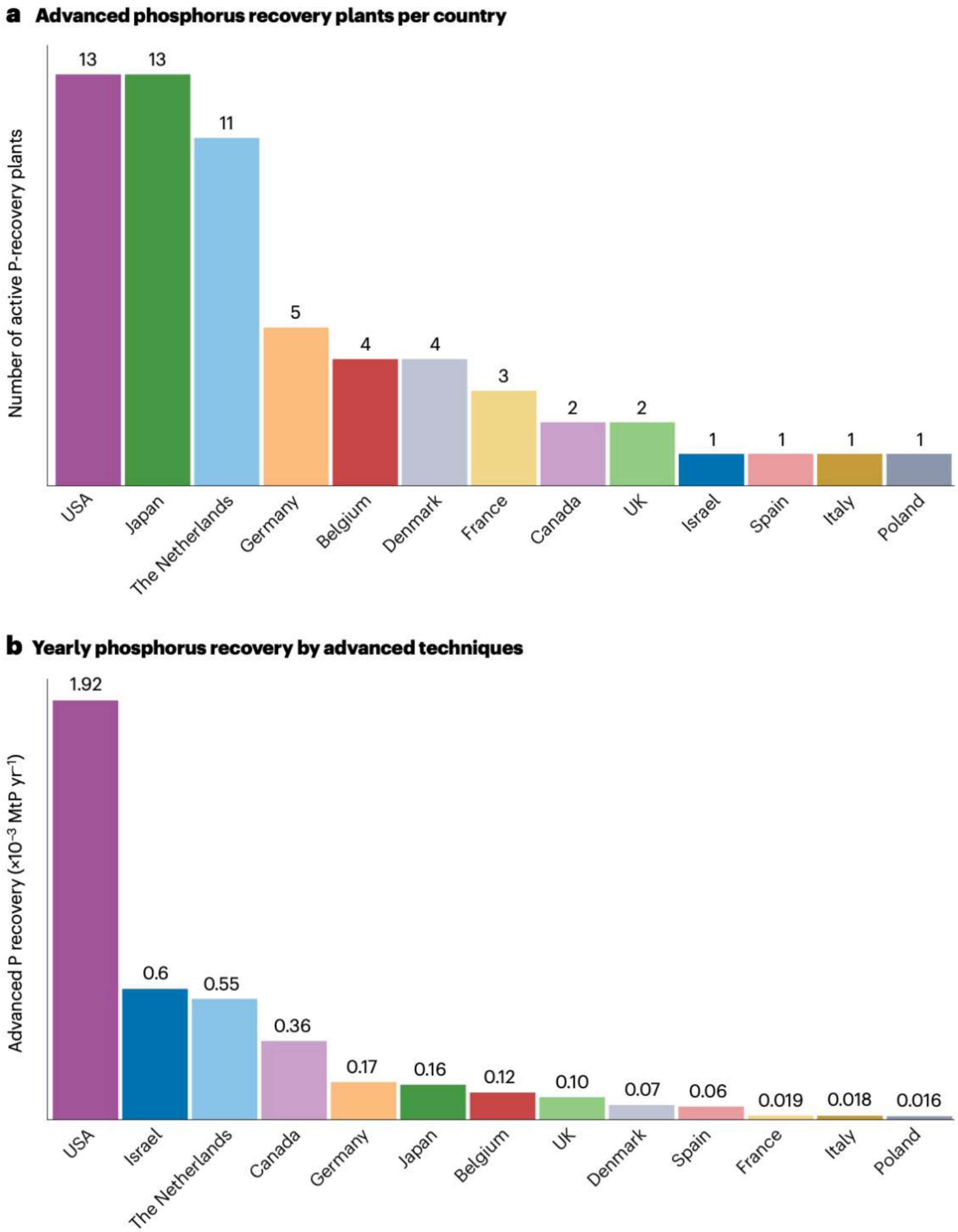

Figure 3. Global distribution and recovery amounts of advanced phosphorus recovery plants. a, The number and distribution of operational advanced phosphorus recovery facilities in each country in 2023. b, The annual total phosphorus output by the 61 advanced phosphorus recycling plants that reported their yields in 2016 (ESPP, 2023). MtP, million tones phosphorus.

2.5 Legacy phosphorus

In agriculture, P input through the application of mineral and organic P fertilizers often exceeds the amount absorbed by crops, resulting in a gradual accumulation of legacy P in the soil (Lou et al., 2018; Solangi et al., 2023). Most legacy P is strongly bound to the soil matrix, primarily to inorganic soil constituents, or is lost to aquatic environments through erosion, run-off or leaching (Lun et al., 2018; Tu et al., 2023), eventually accumulating legacy P in sediment (Kiani et al., 2023; Solangi et al., 2023; Tu et al., 2023), where it often contributes to environmental issues, such as eutrophication. Although legacy P is often not immediately available to plants, physical, chemical and biological pathways exist that can mobilize this P, allowing a portion of legacy P to be taken up by future crops grown in the same field (Gatiboni et al., 2021).

2.5.1 Legacy phosphorus in soil

Globally, excessive fertilizer use between 1965 and 2007 resulted in the accumulation of over 815 Mt of legacy P in the soil (Sattari et al., 2012). The high reliance on P fertilizers is particularly evident in Western Europe, Brazil, North America and Asia, where anthropogenic P accounted for nearly 60% of the available soil P in 2017 (Demay et al., 2023). By contrast, Africa relies on these P inputs for only approximately 30% of the total available P in the soil (J. Zhang et al., 2017). Although P fertilization is projected to rise from 20 MtP in 2023 to 22 MtP in 2025 (U.S. Geological Survey, 2024), only ~30% of the P supplied to crops is absorbed in the year of application, and ~45% becomes legacy P (Pavinato et al., 2020) (the remaining 25% is lost through erosion, run-off and leaching). For example, soil legacy P was ~9 Mt in 2023 (Lun et al., 2018). In the EU, excessive fertilizer use has led to an accumulation of ~222 Mt of legacy P in topsoil (Panagos, Köningner, et al., 2022), and legacy P accumulated in agricultural soil could reach ~107 Mt by 2050 in Brazil (Pavinato et al., 2020).

Efforts have been made to utilize or ‘mine’ soil legacy P. One promising approach is cover cropping with species that are capable of accessing less-available forms of P (Hallama et al., 2019) through adaptations such as altered root structures and architectures (Lambers et al., 2006), high rhizosphere acidification capacity (Y. Zhu et al., 2005) and increased root exudation of compounds such as phosphatases and carboxylates that enhance P mobilization and availability (Richardson et al., 2011). The biomass of cover crops is then incorporated into the soil, where it releases bioavailable P that can be used by subsequent crops (Hallama et al., 2019). Bioengineering and breeding crop varieties with higher P mobilization potential are also being investigated (Jha et al., 2023; Ojeda-Rivera et al., 2022). However, these approaches are complex and heavily influenced by specific environmental conditions and soil P status (Oburger et al., 2011), which can limit their reliability. Furthermore, newly dissolved P resulting from the effects of freeze–thaw cycles on cover crop biomass can reach nearby waterways (Carver et al., 2022).

Biostimulants are another promising strategy for mobilizing legacy P (J. Zhu et al., 2018). These substances alter the soil microbiota or promote P scavenging by roots to enhance crop P uptake (Ogunsanya et al., 2022). Biostimulants work by mobilizing relatively stable forms of P into plant-available forms, which can considerably increase the efficiency of P use in agricultural systems, particularly in soil with high legacy P levels.

2.5.2 Legacy phosphorus in sediment

Historical P loading from wastewater, run-off and erosion from agricultural soil has led to the accumulation of ~2,686 Mt of legacy P in aquatic sediment, with an estimated current accumulation rate of 1.5 MtP yr^{-1} (Tu et al., 2023). Waterbody management practices focus on maintaining or restoring water quality by immobilizing P in the sediment, which is typically achieved by adding P-binding agents or aerating lake water to promote the binding of P to iron in sediment (Kirol et al., 2024). Although these practices reduce P reactivity and mitigate potential environmental damage, they do not facilitate P recycling. By contrast, reusing lake sediment (Kiani et al., 2023) or filtering and recycling P-rich lake water would help to achieve P circularity.

Despite the substantial P stock in sediment, exploiting this resource by recycling P faces notable challenges that hinder its feasibility. The P recovery process involves complex and costly steps, including dredging, flocculation and dewatering of sediment (Simoni et al., 2024). Then, the treated sediment must be transported and applied to agricultural soil, which adds new layers of complexity, expense and risk if the sediment contains contaminants. Although using P in sediment holds promise and could contribute to P supply, more research and technological development are needed to overcome these hurdles.

2.6 Barriers to phosphorus recycling

In Europe, China, Japan and North America, financial capacity is adequate for developing and deploying efficient P recycling technologies (Hosseinian et al., 2023; X. Liu et al., 2023; Powers et al., 2019; Walsh et al., 2023). However, numerous interconnected barriers pose considerable challenges to implementing efficient P recycling. These obstacles also increase reliance on imported mineral phosphates, exacerbating geopolitical and environmental vulnerabilities.

Acknowledging the interconnectedness of these barriers underscores the need for transdisciplinary approaches, where diverse sectors (including academia, industry, consumers, farmers and others) collaborate to develop comprehensive and sustainable solutions for P management and recycling.

2.6.1 Technological barriers

Developing P-recycling capabilities is the most apparent technological challenge. The heterogeneity of secondary P resources complicates technology transfer between waste streams. For example, P-recovery technologies designed for wastewater are not easily adaptable to recovering P from manure or lake sediment owing to different requirements regarding removal of pollutants, pathogens and contaminants (Hollas et al., 2021). Furthermore, some recycling methods can separate contaminants from waste but often result in poor P recovery or reduced P availability. The best available techniques for recovering P, such as incineration, pyrolysis, thermochemical processing and electrodialysis, are typically chemically intensive, operationally complex and costly (Directorate-General for Environment, 2022; Huygens D et al., 2022), and all yield products that have downsides, notably low P bioavailability without further chemical processing (Egle et al., 2016), and the continued presence of some heavy metals and other pollutants.

Nonetheless, various P recovery technologies are available and are increasingly being implemented at full scale (ESPP, 2023). The challenge lies in balancing treatment costs with performance. Ensuring high P recovery rates from secondary resources while producing clean, safe and highly efficient fertilizers remain difficult to optimize simultaneously.

A less obvious technological barrier relates to the production of knowledge. Standardized analytical methods for assessing P availability do not effectively characterize highly complex, heterogeneous products (Duboc et al., 2022; Sichler, Becker, et al., 2022). Worse yet, no standardized method for P speciation (the chemical forms in which P occurs) in recovered products currently exists, although X-ray diffraction is often used. Although effective for identifying mineral P forms, this method is unsuitable for detecting organic P, is inherently qualitative and fails to provide accurate results when mineral phases are poorly crystalline or amorphous.

These data are crucial for generating, evaluating and comparing the efficiency of recovered P sources, relative to conventional fertilizers (Duboc et al., 2022; Hernandez-Mora et al., 2024). Furthermore, the heterogeneity and complex chemical composition of secondary P resources hinder their evaluation as fertilizer alternatives. For example, in the Netherlands, increasing amounts of macerated food waste are being diverted to sewage systems, contributing to mixed flows being treated in wastewater treatment plants (Smit et al., 2015).

2.6.2 Knowledge barriers

These technical barriers result in knowledge barriers: gaps in understanding of the dynamics of recycled materials in the environment and their efficiency as fertilizers. For example, lack of knowledge of P availability and speciation hinders developers of models for improved P management and for regulators who need these data to proceed sensibly. Recycled products often have lower water solubility and slower P dissolution dynamics than conventional mineral P fertilizers, but can release similar amounts of P over longer periods (Talboys et al., 2016) and achieve similar agronomic efficiency (Deinert et al., 2023; Dox et al., 2022). However, the long-term effects of continuous use of recycled fertilizers are still mostly unknown. The characteristics, production process, application method and environmental risks of the various recycled P sources are not sufficiently understood to anticipate how they will affect fertilizer performance and the environment over time (Devault et al., 2025; Kratz et al., 2019). Therefore, understanding the dissolution kinetics and overall behaviour of recycled fertilizers in different environments is essential for modelling their transport in fields and watersheds.

In addition, the complex interplay between soil particles (such as Fe and Al oxyhydroxides and clay minerals) and the P in fertilizers is not sufficiently understood. P phytoavailability and legacy P accumulation rates vary considerably across systems. Although slower fertilizer solubilization can be advantageous in acidic soil, improving crop uptake and reducing leaching and run-off, P release from biochar and struvite, for example, is considerably delayed or reduced in alkaline soil, and therefore is potentially too slow for optimal plant growth (Buss, 2021; Hertzberger et al., 2020). Furthermore, soils with high P sorption capacity, such as those rich in iron and aluminium oxides, can tightly bind P, increasing soil legacy P, whereas sandy or organic-rich soils often retain less fertilizer-derived P, promoting higher bioavailability.

Current understanding of mineral-bound P species is uneven. Whereas iron phosphates have received increasing attention over the past decade, particularly those precipitated from sewage sludge, research on aluminium-P forms, which is also highly utilized for P precipitation in wastewater treatment plants, remains underexplored. This gap limits the development of P-recovery processes from aluminium-rich waste streams and further complicates efforts to assess the potential and limitations of different recycled P sources in a comparative, system-wide manner.

These knowledge gaps complicate the accurate measurement of P circularity. For this reason, the true economic, environmental and climate-related effects of failing to recycle P are still not fully understood. These uncertainties impede wider adoption of recycled P alternatives, even by early adopters (ESPP, 2023). Without sufficient production, potential users cannot fully test recycled P sources in real production scenarios, which raises concerns about their effectiveness in agricultural systems. This uncertainty lowers farmer demand, which lowers profit expectations for producers, thereby further hindering effective P recycling (Lam et al., 2020; F. Zhu et al., 2023).

2.6.3 Logistics-related barriers

Logistical barriers refer to obstacles that impede the collection, handling, processing and redistribution of waste streams and recycled P. A notable logistical challenge is the distance between the sources of recycled P and arable land where it is needed. For example, densely populated areas generate substantial amounts of P-rich wastewater, but these regions are frequently far from agricultural lands that require P inputs (Panagos, Köningner, et al., 2022). For example, in the Netherlands, sewage sludge application on farmland has been banned since the 1990s owing to concerns about contaminants. Following an incineration capacity shortfall in 2020, the Netherlands exported 27.5 kt of sewage sludge to the UK, one of the few nearby countries that still permit land application of treated sludge (Harvey, 2020). Besides being logistically difficult, this trade was economically impractical, resulting in a financial loss.

Regional generation and distribution imbalances are an issue with the use of manure (Devault et al., 2025; Flynn et al., 2023; Köninger et al., 2021). The high volume and moisture content (85–95%) of manure create logistical challenges for its handling, storage and long-distance transport (Ghimire et al., 2021; Lessmann et al., 2023; Tonini et al., 2019). The water content of manure can be reduced, thereby increasing P concentration, but logistical shortcomings continue to limit the accessability of manure for P recycling (J. Li et al., 2021; Teenstra et al., 2014). Existing techniques such as thermal drying can reduce moisture content to 10–15% but are expensive and energy intensive (Khodadadi et al., 2023). The use of more accessible, non-thermal techniques results in a product with 65–75% water content, which is still too high for efficient, cost-effective application and transportation. In Sweden, if fertilizer prices remain stable and recycling processes are unchanged, transportation costs would need to be reduced by 73% for manure recycling to be cost-effective (Akram et al., 2019).

Last, recycled P fertilizers are cumbersome, complicating their application (Lessmann et al., 2023; Panday et al., 2024). Recycled P fertilizers are often bulky or dusty, which makes them difficult to handle and requires specialized and/or multiple machines to manage large volumes. For example, although food waste-based compost and digestate can improve soil structure and organic carbon content, their P content is low (~0.4% P on a dry matter basis (Hosseinian et al., 2023; Otles et al., 2015)). As more concentrated P fertilizer options exist, farmers resist extra investments that would facilitate the use of more bulky, recycled options. In addition to P, recycled fertilizers can contain variable nitrogen, potassium and micronutrient content, and this variability, coupled with differing nutrient-release dynamics and logistical costs, tends to dissuade farmers from using recycled products (Case et al., 2017).

2.6.4 Economic and financial barriers

The economic feasibility of safely recovering and using P from secondary sources is another barrier to P recycling (Grieger et al., 2024). High initial investments, elevated production costs and uncertain returns deter the implementation of P-recovery technologies (F. Zhu et al., 2023). The financial magnitude of full-scale P recycling from sewage sludge ash into technical-grade phosphoric acid (75% H_3PO_4, containing 23% P) has been demonstrated in Switzerland. As of 2023, the projected capital expenditure for a facility producing 40 kt of technical-grade phosphoric acid annually is ~US$190 million, with operational costs of ~US$29.5 million yr^{-1} to produce 12 kt of phosphoric acid annually (2,760 tons P(tP) yr^{-1}). Thus, operational costs are ~US$2,460 t^{-1} of phosphoric acid and annualized capital expenditure is ~US$9.3 million (~US$775 t^{-1} of phosphoric acid) (Schlumberger, 2023). At the time, the market price of phosphoric acid was below US$1,100 t^{-1}, about three times less than the production cost of recycled phosphoric acid (FOB Rotterdam, 2024; Schlumberger, 2023).

The higher cost of recycled P fertilizers is related to the physicochemical characteristics of the secondary materials, the smaller operation size (economy of scale) and the increased complexity of most yields a final product that is too expensive for most farmers compared with raw manure (Devault et al., 2025). Similarly, fertilizers produced with struvite precipitation technologies can be 2–14-fold more costly than those derived from phosphate rock (Egle et al., 2016; Maaß et al., 2014; Uzkurt Kaljunen et al., 2022). Most attempts at struvite precipitation (related to clogged pipes from uncontrolled struvite crystallization) have been motivated by an interest in reducing maintenance costs, not in producing fertilizers (Mudragada et al., 2014; Siciliano et al., 2020). The higher cost of recycled fertilizers compared with those from conventional sources and their uncertain benefits in terms of crop yield can considerably discourage farmers from using recycled materials (Maaß et al., 2014).

P recycling is also markedly influenced by global economic disparities. For farmers, who often even struggle with the cost of conventional fertilizers, investing in recycled P sources is risky, with an uncertain and potentially delayed return on investment (Krüger & Adam, 2017; Rice & Vos, 2024). This scenario leads to reluctance in committing resources to adopt recycled alternatives, hindering P-recycling implementation in developing regions (Jupp et al., 2021; van der Kooij et al., 2020).

2.6.5 Societal barriers

Societal barriers encompass poor food planning, misinterpretation of ‘best-before’ dates for perishable foods and over-purchasing, all of which are core contributors to household waste (Chia et al., 2024; Schanes et al., 2018). Furthermore, perceptions of recycled fertilizers, including their safety, can lead to acceptance-hindering stigmas, including concerns about contaminants and pathogens, as well as ‘yuck factors’ such as odour (Marks et al., 2008; Ricart et al., 2019). These perceptions can discourage farmers from embracing recycled fertilizers and deter consumers from buying produce grown with them (Grieger et al., 2024; Martin-Ortega et al., 2022). In addition, farmers often resist shifting from established practices that they know are effective (Case et al., 2017). They know that recycled fertilizers release nutrients more slowly but are uncertain about the implications of this difference. Their hesitance is exacerbated by a lack of expert advice and guidance that is tailored to their specific situations, leaving many farmers unsure about the most effective use of these products (X. Zhang et al., 2020).

2.6.6 Regulatory barriers

Regulatory and legal barriers to P recycling are related to policies and legislation that fail to effectively support P recycling at various scales. Issues range from poorly focused regulations and guidelines for P use, management and recycling, the absence of quality assurance procedures, inadequate governmental incentives and lack of collaborative goal setting (Tyllianakis et al., 2023), to the unexpected and adverse effects on domestic P recycling of regulation designed to address 'unfair' agricultural trade policies at the international level (Cardwell, 2023). All of these issues hinder the development of strategies and clear frameworks for P recycling globally (Brownlie et al., 2021).

Market access to recycled fertilizers is often impeded by existing subsidies for mineral P fertilizers, but incentives to adopt recycled products are also insufficient, perpetuating their low competitiveness (Metson et al., 2022). These policies discourage the use of recycled products and limit their global trade potential, further stalling efforts to create a circular economy for P recycling. One notable regulatory barrier is the lack of authorization for using recycled fertilizers in agriculture. For example, countries such as China, Japan and the USA have made technological advances in P recovery from wastewater, but lack policies that actively support or require P recovery from the wastewater sector (Carrillo et al., 2024). A major reason is quality concerns regarding products that are feared to contain heavy metals, pathogens, microplastics and toxic organic compounds, such as per- and polyfluoroalkyl substances. These fears lead to regulatory and market barriers (Arcas-Pilz et al., 2023). Only a few countries worldwide (that is, Germany (AbfKlärV, 2017), Austria (Sustainability & Tourism, 2017) and Switzerland (Council, 2015)) have implemented regulatory mandates for P recovery from sewage sludge (Sichler, Adam, et al., 2022). This regulatory lag limits the widespread adoption of P-recovery technologies in many regions.

Similarly, up until the 2020s, legal approval for struvite to be used as a fertilizer was not forthcoming, but this use is gradually being accepted globally. In the EU, legislation is following this trend: an amendment of the Fertilising Products Regulation (EC2019/1009) (European Comission, 2019) now enables the use of struvite as a fertilizer. However, other major P-rich secondary sources still lack approval. An example is Category 1 animal by-products (animal parts suspected or confirmed to be infected by biological hazards), which can have high P concentrations, but are not currently authorized under the EU's Fertilising Products Regulation owing to health concerns. For example, there is a potential risk of contamination with prions that cause bovine spongiform encephalopathy, and despite some thermal treatments and downstream acidulation showing reasonable efficiency in eliminating such risks (Sakudo et al., 2021), other studies are less conclusive (Allende et al., 2025) and the techniques lack validation at scale. More complex certification processes for these products place an additional burden on producers, who must ensure compliance to gain market access, and ultimately discourage P recycling (ESPP, 2023).

There is a lack of harmony between the regulations and guidelines for different P-related sectors. Numerous sector-specific regulations exists for P management but there is no overarching governance framework (Garske et al., 2020). For example, in the EU, the Fertilising Products Regulation (EC2019/1009) now includes recovered P fertilizers, but frameworks that regulate secondary P resources, such as the Urban Wastewater Treatment Directive (91/271/EEC) (Commission, 2019), offer no guidelines about recovered P. Using manure or sewage sludge, farmers often apply the maximum nitrogen dose allowed by the EU Nitrates Directive (91/676/EC) (Asai et al., 2014; European Comission, 1991), potentially resulting in P overapplication (Köninger et al., 2021). Although many EU member states provide P application guidelines, relatively few have legislation focused on the use of P fertilizers (Steinfurth et al., 2022). This sectoral approach perpetuates technocratic objectives focused on narrow goals (Kalpakchiev et al., 2023), relying heavily on command-and-control instruments that are weakly enforced. These hindrances result in a fragmented policy landscape that impedes the systemic adoption of P recycling practices (Garske & Ekardt, 2021).

The fragmentation in legislation is further exacerbated by stakeholders operating in disciplinary silos with a narrow focus on their specific concerns rather than collaborating to establish a holistic framework for P management (Deviney et al., 2023). For example, stakeholders concerned with contaminants in waste-derived fertilizers, such as environmental advocacy groups, or the general public concerned with the ‘yuck factor’ of these fertilizers, tend to advocate for strict regulations regarding P recycling without fully considering the economic challenges faced by farmers and wastewater companies (Grieger et al., 2024). These concerns highlight the broader challenge of determining when such materials can be considered safe to use, which complicates regulation and policymaking, a key issue in the ongoing ‘End-of-Waste’ debate. This dilemma points to the need for shared knowledge, open dialogue and a community-driven willingness to compromise in pursuit of more balanced and workable policies.

2.7 Strategies for moving forwards

2.7.1 Improving phosphorus flows

Addressing the global P challenge requires a multifaceted, holistic approach that harnesses the expertise of individual disciplines (Kalpakchiev et al., 2023). The first pathway to better P management is to reduce the volume of secondary P. In agriculture, this approach includes developing precision farming techniques that increase application efficiency and reduce run-off to water bodies (Pätzold et al., 2020). In addition, better manure recycling and the integration of livestock and crop production are needed to use manure more efficiently as a P fertilizer. In urban contexts, this approach includes decreasing food waste and broadening the collection and treatment of wastewater to recover P and reintroduce it into the cycle. P recovery through waste treatment should be expanded globally, with investments in infrastructure that support P recycling from agricultural, industrial and urban waste (Boer et al., 2018).

Increasing the recovery from secondary P resources must be complemented by reducing demand for P resources. Transformative changes in human diets and consumption patterns are essential to mitigate the effects of P-intensive food production. Over the past 50 years, per capita P footprints have surged owing to dietary shifts, primarily rising consumption of meat, which now comprises 72% of the global P footprint. This shift has driven a 38% increase in global P demand between 1961 and 2007, with substantial variations across countries (Metson et al., 2012). Reducing meat consumption, especially in countries with high P footprints, could substantially decrease P demand (Metson et al., 2012). This change would both help conserve finite P resources and reduce the risk of eutrophication, thus aligning with broader environmental and health sustainability goals. In addition to dietary changes, improving P use efficiency and enhancing recycling at each stage of food production are crucial complementary strategies (Metson et al., 2012).

In addition to improved production, the challenges posed by the damage done by P that is lost to the environment remain. Technologies are needed to reduce P run-off into water bodies. Lower P losses can be achieved by effective exploitation of legacy P in soil through intensive farming, cover cropping, biostimulant use and efficient P management (for example, fertilizer choice, application timing and doses). Waterbody restoration techniques should focus on recycling legacy P in sediment for use in agriculture. For example, restoring lakes through sediment dredging could provide P while substantially reducing methane emissions (Davidson et al., 2024). Therefore, tapping into P stocks in soil and sediment could help address both the P and climate challenges. Advanced P recycling can also play a major part in reducing both legacy P accumulation and losses to water bodies. Widespread P recycling would increase the market availability of less water-soluble fertilizers (such as struvite, (bio)chars and sludge ashes), decreasing reliance on water-soluble fertilizers that are prone to run-off (Sohoulande et al., 2023). In some cropping systems, this shift could better align P release with plant demand, reducing P fixation in soil and losses to water bodies (Withers et al., 2014).

Improving P use efficiency in animal feed can substantially reduce P losses across the agricultural system. One well-established approach is the addition of enzymes such as phytase, which increase P availability to animals, thereby lowering P additive to feed and therefore total P intake by livestock and reducing P concentrations in manure (Lautrou et al., 2022). Another strategy is to remove excess P from feed ingredients. For example, in the USA, P is removed from the grains of dry distillers used as feed to reduce its P content before consumption (Ruffatto et al., 2023). However, the need for more efficient use of manure remains. Once manure is produced, recovering P from digestate can further improve overall nutrient recycling (Sajjad et al., 2024). Advanced P recovery from manure often involves solubilizing and precipitating P compounds such as calcium phosphate and struvite (Lorick et al., 2020; Schott et al., 2023). Emerging methods such as vivianite separation, vacuum evaporation, membrane filtration and ion exchange are promising (Sajjad et al., 2024) but not yet fully developed or widely implemented.

2.7.2 Policy and pricing instruments

Prescriptive regulations to reduce harmful practices, mandate P recovery (Hukari et al., 2016) and enforce the use of recovered products are widely advocated (Devault et al., 2025; Egle et al., 2016; X. Liu et al., 2023) but not often implemented. This hesitance has spurred calls for broader approaches, including price-based policy instruments such as auction or tender systems that allocate public funds to support environmental services, and quantity-based mechanisms, including offset programmes (Rolfe & Windle, 2011). The major challenge for any price-based incentive scheme designed to change P use at the country level is preparing for and responding to unexpected effects from World Trade Organization (WTO) rulings regarding food production for export.

Under the rules of the WTO Agreement on Agriculture, pricing policies interpreted as 'market-distorting' are prohibited. Government policies that manipulate the price of P fertilizers might fit this description. Fortunately, exemptions are possible for price-based incentive schemes that do not exceed specific limits, or have "no or at most minimal trade-distorting effects on production" and do "not have the effect of providing price support to farmers" (World Trade Organization, 2022). And, to the extent that any price incentive scheme for recycling P could be part of a 'clearly defined' governmental environmental programme, the amount paid to farmers must be "limited to the extra costs or loss of income involved in complying with the government programme" (Smith, 2015). However, the risk of an adverse WTO ruling remains.

Cap-and-trade systems for P contained in manure (as for CO_2 emissions) have also been suggested for improving P management (Garske & Ekardt, 2021) and have even been implemented in the Netherlands (European Commision, 2017). However, this approach can be problematic because watersheds and water bodies are affected by local pollution. Therefore, unlike CO_2 emissions, choosing where to cap and trade becomes crucial, especially in light of the need for improved manure management and transportation logistics. Regulations should support safe, new technologies and recovered P products in agriculture and industry, despite intersectoral legal difficulties. For example, the EU's Carbon Border Adjustment Mechanism attempts to address regulatory issues regarding CO_2 emissions on imported goods, but questions remain about the feasibility of the implementation and compatibility of the Carbon Border Adjustment Mechanism with WTO trade rules. Alternatively, subsidy reforms, such as reducing mineral fertilizer subsidies (as in China (Wu et al., 2024)) or transforming agricultural subsidies (as in the EU and USA) can encourage the use of recovered fertilizers and sustainable practices, shifting subsidies towards improved P management, provided that challenges around subsidy design to ensure compatibility with WTO trade rules can be overcome (Heyl et al., 2022; Hukari et al., 2016).

Large, centralized facilities could take advantage of economies of scale to make P recycling economically viable in areas that are highly urbanized or have dense livestock production (Bagheri et al., 2024). Such facilities optimize labour, energy and raw material use, reducing operational costs and maximizing output efficiency. However, centralization comes with its own set of financial and operational risks, such as supply chain interruptions and regulatory compliance issues, as well as limited ability to adapt to local market conditions (Bagheri et al., 2024). By contrast, decentralized, small-scale solutions such as on-farm urine sterilization or manure processing can be more suitable in rural or low-income settings where nutrient demand and supply are more locally balanced and transport distances are shorter. This contrasting suitability is due to the high capital investment required for constructing and maintaining large-scale facilities, which can be a notable financial burden that affects the decision-making of implementing P recycling.

2.7.3 Transdisciplinarity in phosphorus recycling

Interdisciplinary frameworks and decision-support tools are vital for developing comprehensive P sustainability strategies. The EU-centred 5R phosphorus stewardship framework offers a broad model that includes realigning P inputs, reducing losses, recycling secondary resources, recovering P from waste and redefining food systems through demand shifts (Withers et al., 2015). Technologies and practices must achieve both technical and societal acceptance, addressing both community effects and agronomic needs (Martin-Ortega, 2023). Similarly, the 'net-zero phsophorus cities' framework emphasizes the role of urban areas in creating circular P economies by capturing wastewater P (Metson et al., 2022). These frameworks could be improved with better empirical data (Daniel et al., 2022; Leahey et al., 2017; L. Liu et al., 2023) and holistic integration of disciplinary perspectives.

We call for a transdisciplinary approach that incorporates non-academic stakeholders to ensure full co-production of strategies. Collaboration must transcend individual interests and focus on shared goals, such as reducing run-off, improving nutrient-use efficiency and promoting recycling (Martin-Ortega et al., 2022). Transdisciplinarity enables a holistic understanding of the P cycle by integrating insights from agriculture, waste, environment, economics and policy sectors. Although transdisciplinarity faces challenges such as securing long-term funding, time-consuming coordination and communication among diverse stakeholders (Jama-Rodzeńska et al., 2021; Martin-Ortega et al., 2022), it fosters innovative solutions for P recycling and pollution reduction, ensuring that strategies are practical and sustainable. Transdisciplinary efforts must inform policies that address P management complexities from local to global levels (Garske & Ekardt, 2021; Kalpakchiev et al., 2023).

P-focused organizations, such as the Global Phosphorus Research Initiative, the European Sustainable Phosphorus Platform, the US Sustainable Phosphorus Alliance, the Australian

Sustainable Phosphorus Futures and the Phosphorus Industry Development Organisation of Japan, have a key role in coordinating efforts by connecting stakeholders and facilitating P-recovery strategies (Lyon et al., 2020). These agencies should serve as 'mediators' in stakeholder relationships. In regions where they do not exist, efforts should focus on engaging stakeholders beyond sectoral boundaries, thus paving the way for technological scaling, supportive policies and regulatory frameworks (Martin-Ortega et al., 2022) while ensuring a transition to circular P economies [58].

Transdisciplinary research must promote broad and inclusive participation, ensuring that diverse perspectives are represented in decision-making. This approach supports fair processes and fosters more widely acceptable solutions. The potential of the transdisciplinary approach is exemplified by the UK Phosphorus Transformation Strategy (Cordell et al., 2022) that involves extensive stakeholder engagement and has already produced a credible roadmap for multisectoral action across the P value chain, which can be broadened to other regions (Figure 4). Future efforts should foster knowledge exchange to identify stakeholder interests and challenges, create local pathways, assign responsibilities and develop realistic timelines for P recycling targets (Martin-Ortega et al., 2022).

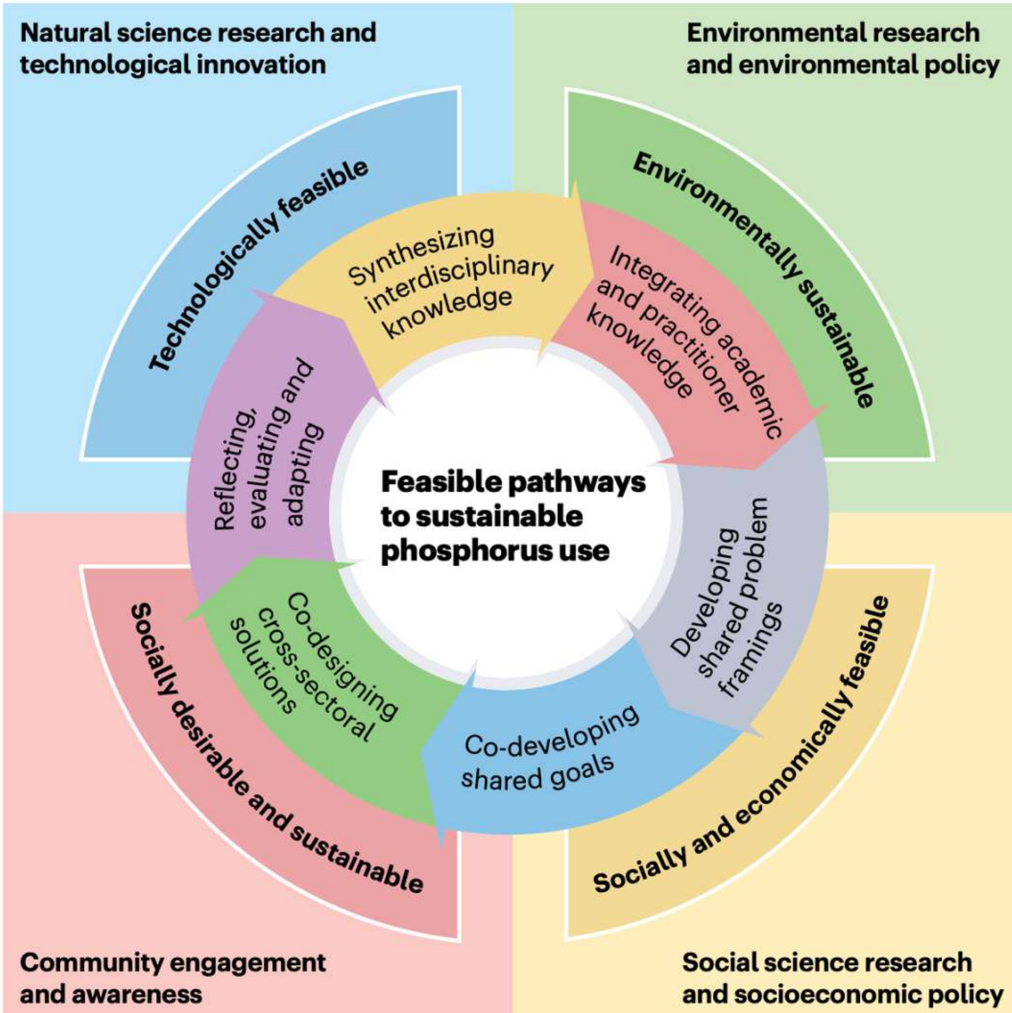

Figure 4. Transdisciplinarity as a tool to overcome fragmentation through integration and stakeholder engagement. The role of a transdisciplinary approach to phosphorus management (Cordell et al., 2022). Fragmentation across research fields and sectors hinders the development of sustainable phosphorus pathways

2.8 Summary and future perspectives

Recycling is an important step towards P circularity, but global adoption remains limited owing to economic, technical, societal, regulatory and political barriers. The technical complexity of P-recycling technologies (Egle et al., 2015), small scale of operations and high production costs make recycled fertilizers considerably more expensive than conventional, rock-derived fertilizers, as shown in Switzerland (Schlumberger, 2023). Recycled P products are also more chemically and physically complex, complicating attempts to understand their agricultural efficiency, effects on human health and long-term environmental safety (Lessmann et al., 2023). Limited production, logistical hurdles in transport and utilization, and uncertain financial returns further discourage adoption, as demonstrated in Sweden (Akram et al., 2019). Meanwhile, narrow, disciplinary stakeholder focus and diverse socioeconomic and environmental realities across regions hinder collaboration and stifle global progress in P recycling (Martin-Ortega et al., 2022).

Despite growing awareness of the importance of P recycling, challenges in accurately determining global P flows and uncertainties around the agronomic efficiency, safety and environmental effects of recycled P fertilizers continue to hamper progress. In addition, diverse agricultural systems and regulatory frameworks further complicate the development of coherent global strategies. This fragmentation extends across the whole P system: researchers, farmers, policymakers, environmental agencies and consumers each prioritize different goals, creating misalignments that undermine coordinated action. Bridging these gaps requires improved communication, transdisciplinary collaboration, and context-specific solutions that balance technical, societal and environmental needs.

Transforming P management requires a holistic approach in which stakeholders prioritize the shared objective of tackling the wicked problem of P circularity. Key goals include minimizing waste generation, reducing P run-off, enhancing nutrient-use efficiency and promoting P recycling efforts. Breaking down barriers across sectors is essential: improved communication, transdisciplinary research and diverse perspectives can lay the foundation for an integrated approach to P circularity. Inclusive policies that address the needs of all stakeholders can drive collective action. Strategic research developments and actionable priorities will support progress over the next decade.

Supporting these efforts requires a concerted push through targeted incentives for sustainable P use; coherent regulation at local, regional and international scales; investments in innovation and research; and the creation of inclusive platforms that facilitate transparent dialogue across sectors. Ultimately, the transition to circular P systems will depend on the ability of stakeholders to break down disciplinary, institutional and geographic silos, fostering collaboration that bridges science, policy and practice. Only then can resilient, sustainable P management strategies be built that are capable of reshaping the current linear supply chains into circular systems for the future.

3. EU-compliant wastewater recycled phosphorus: how much national cereal demand can it meet?

3.1 Abstract

Finding alternative phosphorus sources is imperative to address negative environmental and societal impacts caused by its current inefficient use. However, the direct use of phosphorus in sewage sludge in agriculture is controversial. This paper uses Denmark, Germany, and Spain as case examples to assess relevant legislation and boundary conditions in agricultural production to identify opportunities and barriers for the utilisation of recycled phosphorus from wastewater in agriculture on a regional level. Only five out of 22 phosphorus recycling technologies considered were in full compliance with legislation across all three countries, and these five were then assessed for their potential to supply phosphorus to major crops within countries. We considered the application of technologies across four scenarios: 1) struvite; 2) vivianite as iron supply; 3) vivianite for calcium phosphate precipitation; and 4) ashes for calcium phosphate precipitation. The most suitable scenario identified for Denmark was vivianite for calcium phosphate precipitation, whereas in Spain vivianite as iron supply was identified as most suitable, and ashes for calcium phosphate in Germany. We found that in 2018, the potential phosphorus supply from recycling technologies was on average 0.38, 0.29 and 0.05 kg of phosphorus per capita for Danish, German, and Spanish regions. These quantities could meet 9.1, 21.7, and 10.0 percent of the phosphorus required to produce major cereals in each country (specifically wheat, barley, and rye). Given current legal constraints, wastewater treatment plant connections and agronomic context, the potential contribution of recycled phosphorus is non-negligible in many sub-national regions. Still, to access the full potential of phosphorus circularity clear product specifications and transport and logistics among regions will be necessary.

3.2 Introduction

The inefficient use of mineral phosphorus (P), and organic waste products high in P, has also led to problematic losses of this nutrient into water bodies (Panagos, Köningner, et al., 2022), and to landfills as final sinks (van Dijk et al., 2016). P over enrichment of water bodies (i.e., eutrophication) can lead to biodiversity loss, toxic cyanobacterial species occurrence, and even oxygen depleted zones affecting fisheries, recreation, drinking water, and ecosystem integrity (Preisner, 2023). Such anthropological disturbances of the P biogeochemical cycle to supply food, as well as unequitable access to mineral P sources, or phosphate rock (PR), presents an unprecedented sustainability challenge (Chowdhury et al., 2017; Sandström et al., 2023). Hence, there is interest in finding alternative and more sustainable sources than PR. P-containing waste streams such as municipal wastewater are a viable option, but such streams have yet to be fully integrated into fertiliser markets.

Municipal wastewater is defined as the collected water composed of human excreta and waste originated from toilet use, cleaning, washing and cooking, that contain resources such as nitrogen, P, metals and proteins that could potentially be recovered (Ostermeyer et al., 2022). In the European Union (EU), there still is a large untapped potential for reusing P from wastewater. In 2018, the EU imported 1.26 million t of mineral P and used 1.11 million t of P in agriculture (*FAOSTAT*, 2022). The largest losses have been attributed to the consumption sector (0.66 million t P in 2005; (van Dijk et al., 2016)). The consumption sector is composed of plant and animal-based food and non-food products, such as fibres, tobacco, skins/hides, pet food, detergents, wood and paper (van Dijk et al., 2016). The largest share of the total losses from consumption was communal sewage sludge, a by-product of wastewater treatment, with 34.6% (van Dijk et al., 2016). For this reason, the use of P from wastewater sludge as an alternative source for crop production has been widely studied as one of the promising pathways for substituting PR with P recycling (Kanteraki et al., 2022; Sichler, Adam, et al., 2022).

Currently, direct sewage sludge reuse on cropland is allowed in all EU Member States if it complies with national regulations. EU countries adopted the *Sewage Sludge Use in Agriculture* Directive 86/278/EEC while some countries introduced stricter limits for certain heavy metals (see Table A1) (Gianico et al., 2021). Even though untreated sewage sludge can contain between 0.35% and 1.22% of total P from the total solids, it could also contain pollutants such as heavy metals, organic contaminants, antibiotic-resistant pathogens and microplastics (Egle et al., 2016;

Kanteraki et al., 2022). In Europe, approximately 6 kt of microplastics are added every year to soils from sewage sludge (Kanteraki et al., 2022). In contrast, some recycled P products (e.g., from incinerated sewage sludge) have shown lower heavy metal and no organic contaminant accumulation by fertiliser application, due to elimination during the incineration process and post-treatment, as opposed to direct sewage sludge application (Weissengruber et al., 2018), although incineration requires more energy (Egle et al., 2016). Therefore, possible soil pollution from direct sewage sludge reuse in agriculture may be avoided using more processed recycled P products.

Among the EU countries, the conditions for both legislation and infrastructure can greatly differ, not to mention P use in agriculture, which depends on soil properties and type of products. To explore these conditions more fully, we selected three case study countries: Denmark, Germany, and Spain. All three countries are subject to EU regulations but differ in terms of national laws, infrastructure, and agricultural production. Denmark has more stringent P discharge limits in legislation. Germany has a more stringent regulation when reusing sewage sludge for agriculture and has been implementing monoincineration as a preferred disposal route for its management, whereas Spain and Denmark still present more sludge reuse in agriculture (Collivignarelli et al., 2019; Mannina et al., 2023). We selected these three countries to identify opportunities to increase the utilisation of products derived from P from wastewater in agriculture at a regional level. Specifically, this study aims to address three questions:

i. What are the current legal and infrastructural preconditions for the implementation of wastewater P recycling technologies for eventual use on agricultural soils?
ii. What P recycling technologies are most compatible with current wastewater treatment plant infrastructure and regional soil properties?
iii. What is the share of P demand in agriculture that can be potentially supplied from P recycling technologies?

3.3 Material and methods

We first identified relevant legislation for each selected country and the EU regarding P recycling for agricultural use (see Figure A1). Subsequently, we characterised the P recycling technologies from wastewater according to their P stream within the wastewater treatment process and the final recycled P product. Regarding fertiliser legislation, we evaluated if the P product resulting from each technology, listed in relevant European P platforms (e.g., European Sustainable Phosphorus Platform or ESPP, and Nutrient Management and Nutrient Recovery

Thematic Network or NUTRIMAN), would comply with P content requirements and pollutant limits. The P products that complied with legislation were then selected to represent four implementation scenarios, and we determined the potential P supply of each scenario across each country according to the P inflow of wastewater reported in the Waterbase database from the *Urban Waste Water Treatment Directive* (UWWTD). Moreover, we estimated the potential P demand of selected crops based on fertilisation recommendations for soil P maintenance and build-up specified for each crop and region, depending on their soil pH and P content provided by the Land Use and Coverage Area Frame Survey (LUCAS) topsoil database. Lastly, to calculate the coverage of P requirements for the selected crops by recycled P we divided the potential P supply of the most suitable product by the potential P demand.

3.3.1 Legislation for phosphorus recycling

The relevant legislation that involves the use of recycled P technologies for agriculture is based on how P is managed in wastewater and derived waste streams, and standards that fertilisers must comply with. The legislative map, both national and EU level, is divided into sections of the process (Collection, Treatment, Outlet and Usage), and we summarised their requirements per section (see Table A2). The first two regulate how wastewater is to be collected and treated. The aim of treatment is that the outlet discharges ‘clean’ water, and this process generates sewage sludge, which contains all of the P removed from the wastewater during the process. Both the outlet water and the sewage sludge produced are subject to management regulation. In addition to collection, treatment, and outlet sections, the use of P recycled products is regulated under fertilisers and soil conditioners ordinances.

Wastewater legislation throughout the EU is based on the 1991 of the UWWTD 91/271/EEC with its later 1998 amendment 98/15/EEC, where the objective is to preserve the environment and protect people from risks associated with pollution. In this directive, wastewater from agglomerations of over 2,000 population equivalent (p.e.) must be collected and treated with primary and secondary treatment; agglomerations over 10,000 p.e. must have advanced wastewater treatment for designated areas with a high risk of eutrophication, or sensitive areas (Garrone et al., 2018). Denmark identifies all their surface water bodies as sensitive areas, whereas Germany and Spain only designate some areas as sensitive (Preisner et al., 2020). As a consequence, the discharge P limits are less stringent for certain German and Spanish water surfaces and do not ensure at least 80% P removal (Table A2).

Sewage sludge management plays an important role in P recycling, and the legislation regarding its management and reuse is based on the EU 86/278/EEC *Sewage Sludge Directive* (Hukari et al., 2016). Each country (or even within individual regions) has different thresholds for sludge reuse in agriculture (see Table A1). The use of recycled P products (not sludge) in agriculture is bound to the *Fertilising Products Regulation* (EU) 2019/1009 legislation. This legislation contains minimal P content requirements for phosphate fertilisers and heavy metal limits depending on the type of P fertiliser (see Table A3 and Table A4). The aforementioned legislative requirements in the selected countries and the EU were the guidelines to evaluate the compliance of recycled P products studied.

The use of recycled P products as fertilisers is regulated by the *Fertilising Products Regulation* amendment of 2019, where under Component Material Categories 12 and 13 (also present in the German *Fertiliser Ordinance* of 2012), which included recycled P products from sewage sludge are covered as “precipitated phosphate salts and derivates” (EU 2019/1009; DüMV, 2012). Moreover, precipitated phosphates have been approved to be used in organic farming in the latest amendment of products and substances for use in organic production (EU) 2021/1165 (EU 2021/1165, 2021). Considering the legal requirements, the use of recycled P products in agriculture is a more sustainable alternative to safely supply P.

3.3.2 Phosphorus recycling technologies and products

In the wastewater treatment process (Figure 5), there are several streams where P can be recovered (Egle et al., 2015). In this study we focus on three main streams that contain considerable amounts of P: centrate (blue), sludge digestate (green) and sludge ashes (red, colours refer to Figure 5).

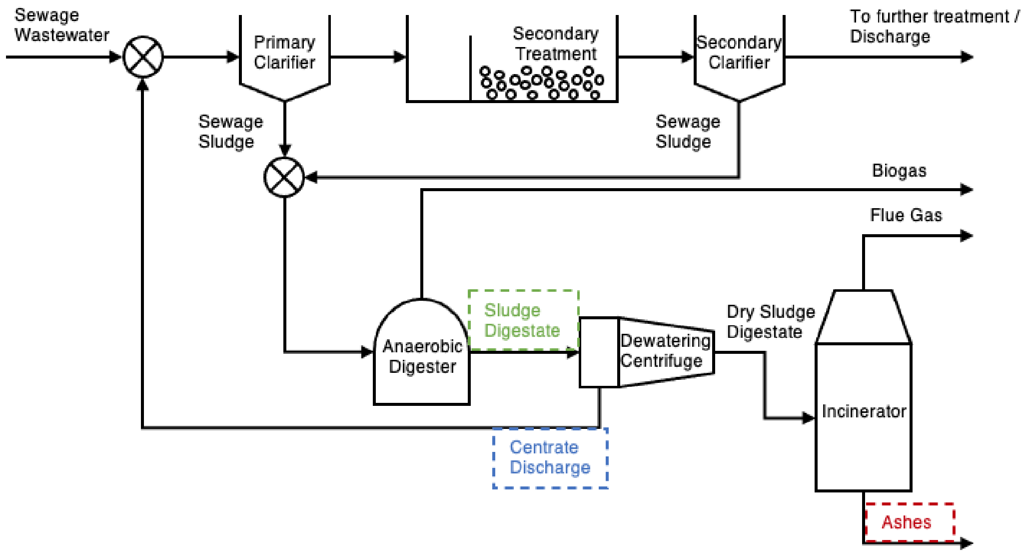

Figure 5. Process outline of a typical wastewater treatment plant (adapted from Egle et al. (2015)).

Typically, after sewage sludge is collected and thickened via sedimentation, it is biologically stabilised via anaerobic digestion, while in parallel biogas is produced through the degradation of organic matter by microorganisms in the absence of oxygen. This stabilised sludge, or sludge digestate, is dewatered to reduce the amount of sludge to be disposed and results in a liquid discharge and solid phase (Egle et al., 2015). The liquid discharge (or centrate discharge in blue – see Figure 5), is recirculated back to the sewage wastewater inlet at the start of the wastewater treatment process. Nevertheless, centrate contains ions that facilitate struvite formation and result in pipeline scaling, which could cause high maintenance cost (Molinos-Senante et al., 2011). On the other hand, the solid phase (or dewatered sewage sludge), due to disposal costs and reduction of methane emissions in landfills, is monoincinerated to ashes when the infrastructure is available.

Technologies to recycle P from different sources within wastewater treatment and their process efficiencies have been widely reviewed (Egle et al., 2015; Santos et al., 2021). P recycling via struvite precipitation from the centrate is one of the most implemented technologies worldwide with around 100 full-scale operations (Jupp et al., 2021). One of the main reasons is that there is a considerable economic benefit of preventing struvite formation in pipes, and also can be directly used as a fertiliser. Instead, P precipitation using iron salts results in vivianite formation, however, it has limited use as fertiliser due to low solubility in neutral soil pH. Consequently, vivianite precipitation could lead to two P-recycled products: vivianite as iron phosphate (Fe-P) for direct application to soils prone to chlorosis or calcium phosphate (Ca-P) through alkaline post-treatment (Prot et al., 2019). Lastly, it is possible to recycle P as Ca-P through acid attack and wet chemical treatment of the sludge ashes (Herzel et al., 2016).

To increase the sustainability of fertilisers used in agriculture, P-recycled fertilisers, such as struvite and Ca-P, have been compared to traditional PR products (e.g., triple superphosphate or single superphosphate) in terms of P use efficiency (Raniro et al., 2022). Recycled P products described in this paper are highly insoluble in water, but present high solubility in neutral and alkaline ammonium citrate, which is commonly used to evaluate the fertilising quality of a non-water soluble P product (Herzel et al., 2016). Several studies on P-recycled products have demonstrated slightly lower or similar fertilising efficiency as PR-derived products for different types of crops such as wheat, maize, rye, among others, and soils (Johnston & Richards, 2004; Oliveira et al., 2019). In contrast, the fertilising efficiency of vivianite is limited to specific soils such as calcareous soils (pH >7.5), in which additionally to P it provides iron for its deficiency in plants, or iron chlorosis (Díaz et al., 2010).

Given that the scope of this study was within the EU, the P-recycled products from wastewater found in the catalogue from the ESPP (*European Sustainable Phosphorus Platform*, 2022) and NUTRIMAN (*Nutrient Management and Nutrient Recovery Thematic Network*, 2022) were evaluated according to their product characteristics and their compliance with the EU and national fertiliser legislation. First, the technologies that provided complete information were compared to national and EU fertiliser legislation (heavy metal limits and minimum P content) and characterised according to the generic P compound (e.g., struvite, vivianite, calcium phosphate). Subsequently, for those recycling P products that complied with legislation in Denmark, Germany, Spain and the EU the potential P supply was calculated according to the regional conditions such as wastewater treatment infrastructure, soil properties and agricultural practices (see Table A5).

3.3.3 Conditions for the selected countries

In this study, the regions of Denmark, Germany, and Spain were studied for 5 specific factors that are directly related to the amount of P to be recycled from wastewater, and the P demand for wheat, barley, and rye production reported in 2018 (Table A5):

- Population
- Cropland area
- Soil properties
- Wastewater treatment infrastructure
- Sewage sludge incineration

Population and sewage sludge incineration data were obtained from Eurostat (except for Denmark – "StatBank"), while cropland area data was taken from national accounts: "StatBank" (Denmark); "Ministry of Food and Agriculture – *Bundesanstalt für Landwirtschaft und Ernährung*" (Germany); and "Ministry of Agriculture, Fisheries and Food – *Ministerio de Agricultura, Pesca y Alimentación*" (Spain). The number of wastewater treatment plants (WWTPs) and total P in influent data was obtained through the Waterbase of UWWTD, and soil properties, such as pH and P content in soil (Olsen method), from LUCAS topsoil database (Bundesanstalt für Landwirtschaft und Ernährung, 2019; Eurostat, 2022a, 2022b; Fernandez-Ugalde et al., 2022; Ministerio de Agricultura Pesca y Alimentación, 2022; StatBank Denmark, 2022b, 2022a, 2022c; Urban Waste Water Treatment Directive, 2018).

Only WWTPs from the Waterbase that reported P removal were considered in this study to assess their potential P supply, given that it ensures the availability of necessary P sources for recycling technologies like centrate, digested sludge, or sludge ash. Those plants were assumed to have the infrastructure of a typical WWTP (see Figure 5), and at least 80% of the total P in the influent is removed according to the UWWTD 98/15/EEC.

Although the P inflow of WWTPs in Castilla-La Mancha (Spain) was reported, the mass of the P inflow of 264 plants was between 10 and 1,000 times higher than WWTPs with similar conditions and possibly an anomaly in the database, thus it was assumed to have a standard value of 1.5 g P p.e.$^{-1}$ day^{-1} (Zessner & Lindtner, 2005). It was also the case of several WWTPs with P removal (213 out of 813) of Spain that did not report P inflow and it was assumed the same standard value to assess their potential. Additionally, the Spanish regions of Ceuta and Melilla did not report data at both Waterbase and LUCAS topsoil databases and were consequently excluded from the assessment.

3.3.4 *Scenarios of phosphorus recycling*

From the recycled P technologies in the ESPP and NUTRIMAN catalogues that complied with European and national fertiliser legislation mentioned previously (see section 3.3.3), three recycled P products (struvite, vivianite and Ca-P) were eligible for fertiliser implementation. Moreover, according to the soil pH characteristics of the selected countries, we identified that vivianite could be used directly as Fe-P or as Ca-P fertiliser depending on the soil pH. Consequently, the potential P supply of struvite, vivianite and Ca-P derived into 4 possible scenarios: struvite, vivianite as Fe-P, vivianite as Ca-P and ashes for Ca-P (Table 2).

Table 2. Scenarios for P recycling technologies by source, recycled P product and their efficiency from total P from WWTP influent.

Scenario	P source	P recycling technology	Recycled P product	Recycling Efficiency from wastewater influent ($\eta_{tech,i}$)	Reference
Struvite	Centrate	Chemical precipitation of struvite	Struvite	10-35%	(Egle et al., 2015; Santos et al., 2021)
Vivianite as Fe-P	Digested sludge	Direct use of precipitated vivianite	Vivianite	~60%	(Prot, 2021)
Vivianite as Ca-P	Digested sludge	Alkaline treatment of precipitated vivianite	Calcium phosphate	~54%	(Prot et al., 2019)
Ash for Ca-P	Sludge ash	Thermochemical treatment of sludge ashes	Calcium phosphate	60-85%	(Egle et al., 2015; Santos et al., 2021)

To consider the established scenarios, four assumptions are crucial to estimate the potential supply:

1. Removal of P by chemical precipitation requires the introduction of iron or aluminium salts that precipitate phosphates present in wastewater influent (Prot et al., 2019). P bound to iron or aluminium is not available in the centrate after dewatering digested sludge, and thus would make struvite precipitation unviable. Therefore, the first assumption is that the implementation of vivianite restricts P available for subsequent struvite precipitation, and those were considered as different scenarios.

2. It was also assumed the lowest recycling efficiency from Table 2 to provide a minimum expected P supply with the mentioned P recycling technologies.
3. Additionally, the regional soil pH was also obtained via the LUCAS topsoil database 2018, and for calcareous soils (pH>7.5) vivianite for Fe-P was considered more suitable than vivianite for Ca-P.
4. The percentage of dewatered sewage sludge that is incinerated as a disposal route was assumed to be fully monoincinerated and used to determine the potential P supply for ash-based recycling technologies (ash for Ca-P). The remaining percentage of other disposal routes was used to calculate the potential from the scenarios of struvite, vivianite as Fe-P and vivianite as Ca-P.

Finally, all scenarios were evaluated for each region within the three countries, and the scenario with the highest potential P supply was selected as the most suitable (see Table A5). To obtain a comparing unit between regions, we normalised the potential P supply and demand with the population of 2018.

3.3.5 Phosphorus supply in regions

For each recycling scenario the potential P supply is calculated, and described by Equation 1:

$$m_{P_{supply,i,j}}(kg\ P\ per\ capita) = \frac{n_{i,j}}{n_{total,j}} \frac{m_{P,j}}{u_j} \eta_i \qquad \text{(Equation 1)}$$

Where the potential supply ($m_{P_{supply,i,j}}$) for i scenario and j region from each country expressed in kg P per capita; number of WWTPs per scenario and region ($n_{i,j}$); the total number of WWTPs per region ($n_{total,j}$); annual wastewater P inflow per region expressed in kg P ($m_{P,j}$); the total population of j region (u_j); and recycling efficiency from P influent of WWTPs for each scenario (η_i).

3.3.6 Phosphorus demand in regions

The assessment of the potential P demand was conducted by analysing the fertiliser recommendation for the crops that are the most relevant in terms of cropland, as well as soil properties such as P content (by Olsen method, or Olsen P) and pH. A considerable part of the grain cropland area in Denmark, Germany and Spain is used to produce wheat, barley and rye (between 70-95%). Hence, those crops were considered to analyse a potential P demand that could hypothetically be supplied by P-recycling technologies in the three countries.

One of the objectives of tackling the P challenge, is to increase the P use efficiency in agriculture, although there has been an accumulation of P in agricultural soils (Panagos, Köningner, et al., 2022). It has been estimated that in the EU there is on average a surplus of 0.8 kg P ha^{-1} year^{-1} in agricultural soils (Panagos, Köningner, et al., 2022), and could lead to water pollution. Therefore, there is a need to implement a more efficient estimation of P fertiliser requirements (P_{rate}), and one possible way is based on two components: build-up and maintenance (Equation 2).

$$P_{rate,j} = P_{exported,j} + 10\, BD\, z\, (P_{Olsen,T} - P_{Olsen,S,j}) \quad \text{(Equation 2)}$$

$$m_{P_{demand,j}}(kg\ P\ per\ capita) = \sum_{i=1}^{N} \frac{A_{i,j}\, P_{rate,j}}{u_j} \quad \text{(Equation 3)}$$

The maintenance component or P exported of j region ($P_{exported,j}$) describes the amount of P taken away when harvesting (Recena et al., 2022). The build-up component is added to the maintenance component when soil P content of j region ($P_{Olsen,S,j}$) following Olsen P extraction in mg kg^{-1} P, is lower than threshold values ($P_{Olsen,T}$). These are the values above which no response in crop yield can be expected if P fertiliser is applied (Delgado et al., 2016; Recena et al., 2022). In case $P_{Olsen,S,j} > P_{Olsen,T}$, it is recommended for P fertilising rate to only consider the maintenance component ($P_{rate,j} = P_{exported,j}$). There was no significant effect of crop type on the $P_{Olsen,T}$ for most cereals studied which showed values between 8.1-17 mg kg^{-1} P, thus the average of 12.6 mg kg^{-1} P was considered (Recena et al., 2022). In addition, the average value for bulk density (BD) was 1.38 t m^{-3} for the LUCAS soil sample, while the soil depth (z) at the LUCAS soil sample was 0.2 m for cropland (Recena et al., 2022).

The potential P demand ($m_{demand\ P,j}$ – see Equation 3) for j region was estimated by the sum of cropland area ($A_{i,j}$) of j region and i crop in ha; the $P_{rate,j}$ of j region according to crop production statistics provided by ministries from each country in 2018; and divided by the population (u_j) of each region to normalise the potential P demand.

3.3.7 Potential phosphorus demand covered by potential phosphorus supply

We assessed the capacity of recycled P products to cover the potential P demand for the selected crops by diving the potential P supply by the potential demand. Nevertheless, regions in Germany such as Berlin, Bremen and Hamburg, as well as Balearic Islands in Spain, did not present information on crop production (Table A5), and it was not possible to establish their potential P demand covered by potential P supply. Additionally, some regions showed over 100% coverage of their potential P demand by P recycling (see Table A6) and could trade with neighbouring regions with lower coverage.

This potential P supply could help increase the potential demand coverage of regions with higher potential P demand per capita. Consequently, the supply potential and demand of the region with a surplus were added to the neighbouring region with the highest potential P demand. As an example, in Spain, the Valencian Community could supply over 100% of its potential P demand, whereas Castile-La Mancha, its neighbouring region with the highest demand per capita, could cover around 5% (Table A7). In this light, both potential P demand and supply were combined to improve the potential P demand covered by recycled P. As opposed to those cases, the potential P supply of regions that did not report crop production information was directly added to the neighbouring region with the highest potential P demand. We estimated a separate coverage assessment, where the regions that had a surplus or did not present crop production information (Figure 8b).

3.3.8 Limitations of the method

In this study we used the mentioned methods to estimate both potential P supply and demand that rely on assumptions, that if not met, might lead to inaccurate quantification of potential for P recycling on a regional level.

For instance, in the case of potential P supply estimation, we identified that some of the available information regarding P inflows in wastewater treatment was abnormal, while sludge disposal routes are only available at a country level. One assumption we considered in the calculation of potential P supply was that the percentage of sludge incineration and agriculture reuse was identical in all regions, and it might lead to inaccurate potential estimations of the technologies studied. Not only the information provided by Eurostat is unclear on the sludge disposal methods but also does not supply detailed data on a regional level, and it is also mentioned by Anderson et al., 2021. Therefore, a more detailed report of sewage sludge amounts and specified disposal routes on a regional level could lead to higher accuracy in the assessment of potential P that could be supplied for agriculture.

In addition, the estimation of potential P demand was based on soil P content using Olsen P reported in the LUCAS topsoil database. Although Olsen P is one of the most widely used soil P tests for non-acidic soils, it does not predict accurately available P on soils with high chemical property variation (Delgado et al., 2010). Thus, different extraction methods should be used to more accurately predict soil P in other types of soils to provide better fertiliser management (Delgado et al., 2010).

3.4 Results & discussion

3.4.1 Phosphorus recycling technologies and regional legislation

Only six P recycling technologies out of the 22 reviewed provided enough information on the physicochemical properties of end products (Table A4), namely Ash2Phos® (Ca-P), ViviMag® (vivianite), Crystal Green® (struvite), PhorWater® (struvite), PAKU® (Ca-P) and AshDec® (ashes) (Table A8). Most technologies listed in the ESPP and NUTRIMAN databases did not explicitly provide product information, like chemical composition, necessary to analyse their compliance with the current fertiliser legislation. The only heavy metal not specified for any of the reviewed technologies was chromium(VI) which has a limit of 2 mg kg^{-1} in the EU, Denmark and Germany, and undetectable by any official method in Spain.

	Ash2Phos - Calcium Phosphate	Crystal Green - Struvite	PhorWater - Struvite	ViviMag - Vivianite	PAKU - Calcium Phosphate	AshDec - Calcium Phosphate	EU 2019/1009
As [mg/kg]	1.4	2	-	2	7	10.6	40
Cd [mg/kg]	**0.1**	0.4	0.4	0.2	1.1	0.1	3
Cr [mg/kg]	1.7	5	3	16	170	90	300*
Cu [mg/kg]	5	-	3	41	400	57	600
Hg [mg/kg]	**0.1**	-	-	-	0.04	**0.1**	1
Ni [mg/kg]	2.5	2	3	9	97	0.9	100
Pb [mg/kg]	3.6	0.2	2	12	15	12.3	120
Zn [mg/kg]	-	2	341	160	870	389	1500

EU upper limit [mg/kg]

0 [mg/kg]

*Figure 6. Specified heavy metal content of the recycled P products in reference to the limits established by the Fertiliser Products Regulation (EU) 2019/1009. Values in bold reported the lower limit measured and values with dash were not specified. For specified zinc of PhorWater – struvite the upper limit of the range was considered (0.4-341 mg of Zn per kg of product). *The*

limit of total chromium is not specified by the Fertiliser Products Regulation (EU) 2019/1009 and the highest limit of Germany and Spain (300 mg of Cr per kg of product) was considered.

In general, the composition of the two struvite technologies was similar, and both had lower heavy metal content compared with the ash-based technologies (see Figure 6). Only PAKU® technology was not in full compliance with the current fertiliser legislation across the three countries and the EU. In the case of this ash-based technology, not only the P content was lower (4.7% total P) than the minimum required by EU fertiliser legislation (7% total P) but also the nickel (Ni) content was higher than permissible limits in Germany, as well as higher zinc (Zn) content than class B labellin(Executive Order Act No 318, 2007; Real Decreto 506/2013, 2013; DüMV, 2012)reto 506/2013, 2013; DüMV, 2012)(see Table A4). Additionally, AshDec® technology presented a Cr and Zn content higher than the class B labelling limit according to the *Fertiliser Products Royal Decree* in Spain, although, it could still be used. This significant variation of most heavy metals, in this case Cd, Cu, Ni, Cr and Zn, is explained by the fact that there is a high variation of substance content among the different wastewater influents depending on the source (Herzel et al., 2016). The variation could happen for P like in the case of PAKU®, which was under the EU minimum P content. In contrast, Ash2Phos® showed a similar P content to the commonly used PR fertiliser triple super phosphate (~20% of total P) (Cabeza et al., 2011).

The five technologies we retained for further investigation (Ash2Phos®, Crystal Green®, PhorWater®, ViviMag® and AshDec®) comply with both EU and national current fertiliser requirements of minimal P content and pollutant limits. Nevertheless, at the moment of evaluating the potential use of recycled P products in agriculture, a complete and standardised characterisation of physicochemical properties is crucial to compare those with commonly used mineral P fertilisers.

More ambitious goals are being set by the EU to achieve the *Sustainable Development Goals* and legislation is expected to change accordingly within the EU (El Wali et al., 2021). At the time of this study, a revision proposal of the UWWTD submitted in October 2022, aims to strengthen the P discharge limits from 1-2 mg P L^{-1} to 0.5 mg P L^{-1} (Proposal for a Directive of the European Parliament and of the Council concerning urban wastewater treatment, 2022). In the case of approval by the European Parliament and Council, the P content in sewage sludge could increase, thus providing more P recycling potential.

Some countries in the EU have also adopted measures to make P recovery mandatory and foster P recycling technologies. In 2017, Germany introduced the *Sewage Sludge Ordinance* amendment which will further restrict sewage sludge reuse in agriculture will be more restricted from 2029 onwards (AbfKlärV, 2017). Furthermore, in 2023 it is expected that WWTPs in Germany report a P-recovery plan for agricultural use.

This has been carving the path for more monoincineration of sewage sludge, which is expected to increase ash-derived technologies potential in Germany (Roskosch & Heidecke, 2018). However, P recycling lacks support in other countries within the EU from the perspective of agricultural and environmental legislation (Garske 2020); more specifically, stricter pollutant limits in mineral fertilisers and sewage sludge, as well as requirements for removal of P from wastewater in usable forms. The unification of such changes in legislation on an EU level would foster a more sustainable and safer use of P to tackle the challenge.

3.4.2 Regional potential phosphorus supply

The most suitable scenario for Denmark was vivianite for calcium phosphate precipitation, whereas in Spain it was vivianite as iron supply, and ashes for calcium phosphate in Germany (Figure 7a). More specifically, vivianite as Ca-P was selected for Denmark because the pH was lower than 7.5 in all regions. In contrast, most regions in Spain had pH higher than 7.5 and were assessed with vivianite as Fe-P, except Galicia, Principality of Asturias, Cantabria, Castile-Leon and Extremadura which had pH lower than 7.5 and vivianite as Ca-P was considered for potential P supply estimation. The potential P supply per capita per region ranged from 3×10^{-4} kg P per capita (Canary Islands, ES) to 0.46 kg P per capita (Central Region, DK), with an average of 0.38, 0.29, and 0.05 kg of P per capita for Danish, German and Spanish regions, respectively (see Table A6).

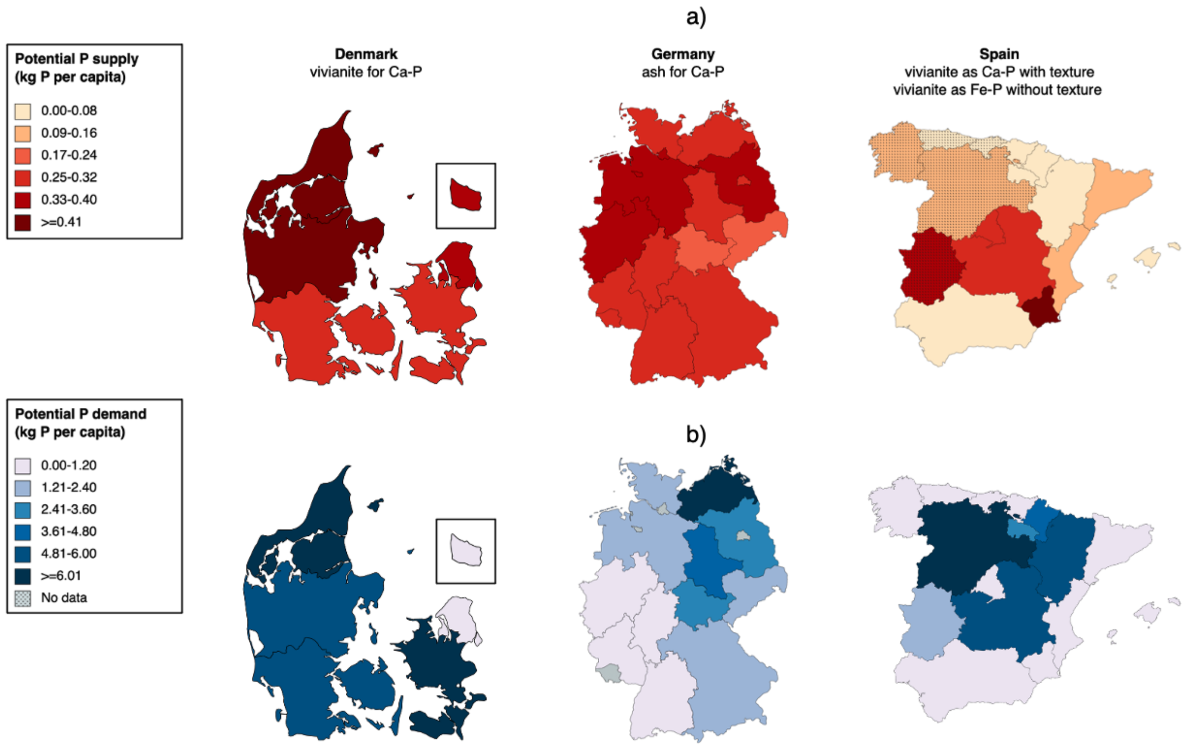

Figure 7. Regional potential P supply (a) and demand (b) in 2018 in case study country regions. Values are expressed as kilograms of P per person (kg P per capita) and each county's potential supply is associated with the highest supply scenario: Denmark as vivianite as Ca-P, Germany as ash for Ca-P, and Spain as vivianite as Ca-P with texture and vivianite as Fe-P without texture – see Table A6 for all values. In the Canary Islands (Spain), the potential supply was 3x10–4 kg P per capita. No soil data was provided in grey regions and was not possible to calculate the potential P demand (including Canary Islands, Spain).

In Denmark alone, the potential P supply was similar across regions and ranged between 0.30-0.46 kg P per capita (Southern and Central regions, respectively). In Germany, the region with the lowest potential P supply was Saxony (0.18 kg P per capita) and the highest Brandenburg (0.40 kg P per capita). Nevertheless, the difference in potential P supply between the lowest (Canary Islands with 3x10^{-4} kg P per capita) and the highest (Region of Murcia with 0.42 kg P per capita) regions in Spain was significantly greater than in the other two countries (Figure 7a). In fact, 13 out of the 17 regions studied from Spain presented a potential P supply inferior to the lowest region in both Germany and Denmark (0.18 kg P per capita, Saxony, DE).

This observed low potential P supply can be explained by the fact that in 2018, the P removal reported from wastewater in Spain was around 34.8%, whereas in Germany and Denmark was higher (95.9% and 85.2%, respectively). According to the data reported in 2017, the percentage of WWTPs that complied with tertiary treatment (described with P and nitrogen removal) was 90.4%, 93.8% and 65.0% for Denmark, Germany and Spain, respectively (Urban Waste Water Treatment Directive, 2021). Moreover, it was estimated that in 2019 Spain had less percentage of the population in small agglomerations (<2,000 inhabitants) from the total population (~30%) than Denmark (~50%) and Germany (~50%) (Pistocchi et al., 2022). Therefore, the potential P supply is directly affected by the number of WWTPs with P removal and not by the amount of P lost in the discharge of small agglomerations as permitted by the UWWTD.

The potential P supply obtained from the countries studied depended on the wastewater collection infrastructure and the sludge disposal route. The 2017 amendment of the *Sewage Sludge Ordinance* in Germany prohibits from 2029 onwards sludge (produced in >100,000 p.e. plants) reuse in agriculture and also requires P recovery. Moreover, in 2032, this prohibition will extend to >50,000 p.e. plants. As a consequence, there is a higher potential P supply by sludge ash-based recycled P products in Germany (Table A5). As opposed in Denmark and Spain, there is a tendency towards sludge reuse in agriculture instead of incineration, which leads to a higher potential P supply from sludge and centrate-based recycled P products (struvite and vivianite both Fe-P and Ca-P) than ash-based products (Ca-P). This way, most of the P content is removed from the sludge, thus leaving its reuse in agriculture less attractive for crop nutrition, albeit the value of organic carbon supply remains. Although, in Denmark and Spain both vivianite scenarios were higher than the ash-based scenario, struvite was significantly lower than both vivianite scenarios, mainly due to low P recycling efficiency from WWTP influent compared with other technologies. In previous studies (Santos et al. 2021), struvite is considered the most promising recycled P product from wastewater or sludge ashes. Our study indicates, however, that with low sludge incineration rates (e.g., <15% in Denmark) struvite from centrate has approximately the same potential P supply (0.06 kg P per capita) as Ca-P.

3.4.3 Regional potential phosphorus demand

From the regions that reported soil P content and pH in the LUCAS topsoil database, the potential P demand per capita average potential P demand was 4.93, 2.38 and 1.18 kg P per capita for Denmark, Germany and Spain, respectively, with a lowest of 9×10^{-5} kg P per capita (Principality of Asturias, Spain) and a highest of 9.94 kg P per capita (Castile-Leon, Spain) (Figure 7b). In Denmark, except for Capital Region, the potential demand was over 5 kg P per capita, whereas only in German Mecklenburg-Vorpommern Region (6.58 kg P per capita) and Spanish Castile-Leon and Aragon (6.47 and 5.07 kg per capita, respectively) was surpassed. German regions such as Berlin, Bremen, Hamburg and Saarland, as well as the Canary Islands in Spain, did not present soil data to evaluate their potential P demand and were marked in grey.

The fertilisation strategy depends on the P exported component, which is based on crop species and yield. In the year 2018, the wheat grain yield was similar for Denmark and Germany (6.36 and 6.67 t ha^{-1}), but considerably lower in Spain (3.90 t ha^{-1}) (Joint Research Centre, 2019). For barley, Germany had the highest grain yield in the same year (5.77 t ha^{-1}), being higher than Denmark (4.53 t ha^{-1}), which in turn, was also higher than Spain (3.51 t ha^{-1}) (Baruth et al., 2019). Grain yields of rye were 5.50 t ha^{-1} in Denmark, 4.30 t ha^{-1} in Germany, and 2.85 t ha^{-1} in Spain. Although Spain presented a lower crop yield in all regions than the other two countries, Castile-Leon region had the highest cereal cropping area (0.88 million ha) (Joint Research Centre, 2019).

In the three countries, the $P_{Olsen,S}$ was higher than $P_{Olsen,T}$, which implied that the P fertiliser requirements for the production of these crops were only for maintenance or compensation of P exported (or P taken away when harvesting). Similar results were also observed (Panagos, Muntwyler, et al., 2022; Recena et al., 2022) when assessing the magnitude of the build-up component for cropland in the EU on a NUTS 3 scale. However, the maintenance strategy could be subjected to further reduction of fertilisation quantities when the ratio $P_{Olsen,S} > 2\,P_{Olsen,T}$ (Delgado et al., 2016). The main reason is that $P_{Olsen,T}$ is a factor that relies on crop type and soil conditions, and depending on the case, if that threshold is surpassed it could lead to higher losses due to erosion (Delgado et al., 2016). As estimated by (Panagos, Köningner, et al., 2022), in the agricultural land of the EU around 374 kt P per year is displaced (or 2 kg ha^{-1} $year^{-1}$ P) and on average 18% of it is displaced to the riverine system and the sea. All regions in Denmark and Germany surpassed the ratio 1:2, while in Spain regions like Basque Community (1:1.4), Aragon (1:1.9), Madrid (1:1.5), Castile-La Mancha (1:1.8), Andalusia (1:1.4) and Murcia (1:1.8) were below (Table A5). Therefore, the potential P demand (Figure 7b) considering the maintenance strategy, could be lower in those cases where $P_{Olsen,S} > 2\,P_{Olsen,T}$, thus making a more efficient use of P considering crop type and soil conditions.

3.4.4 *Percentage of potential phosphorus demand covered by potential phosphorus supply*

The percentage of potential P demand covered only by the potential P supply from P-recycling technologies in wheat, barley, and rye production was on average 9.1%, 23.1% and 10.0% in Denmark, Germany and Spain (Figure 8a), respectively. In Spain alone, six regions (Galicia, Principality of Asturias, Cantabria, Madrid, Valencian Community and Murcia) could supply more than their full P demand (Table A6). In contrast, only Capital Region (Denmark) and North Rhine-Westphalia (Germany) could cover over 40% (74% and 59%, respectively) in both Denmark and Germany. Nonetheless, the Spanish regions capable of supplying the full P demand for selected crops presented a potential P demand lower than 0.15 kg P per capita (Table A6). Also, Capital Region (Denmark) and North Rhine-Westphalia (Germany) had lower potential P demand (0.50 and 0.65 kg P per capita, respectively) than the other regions in both countries.

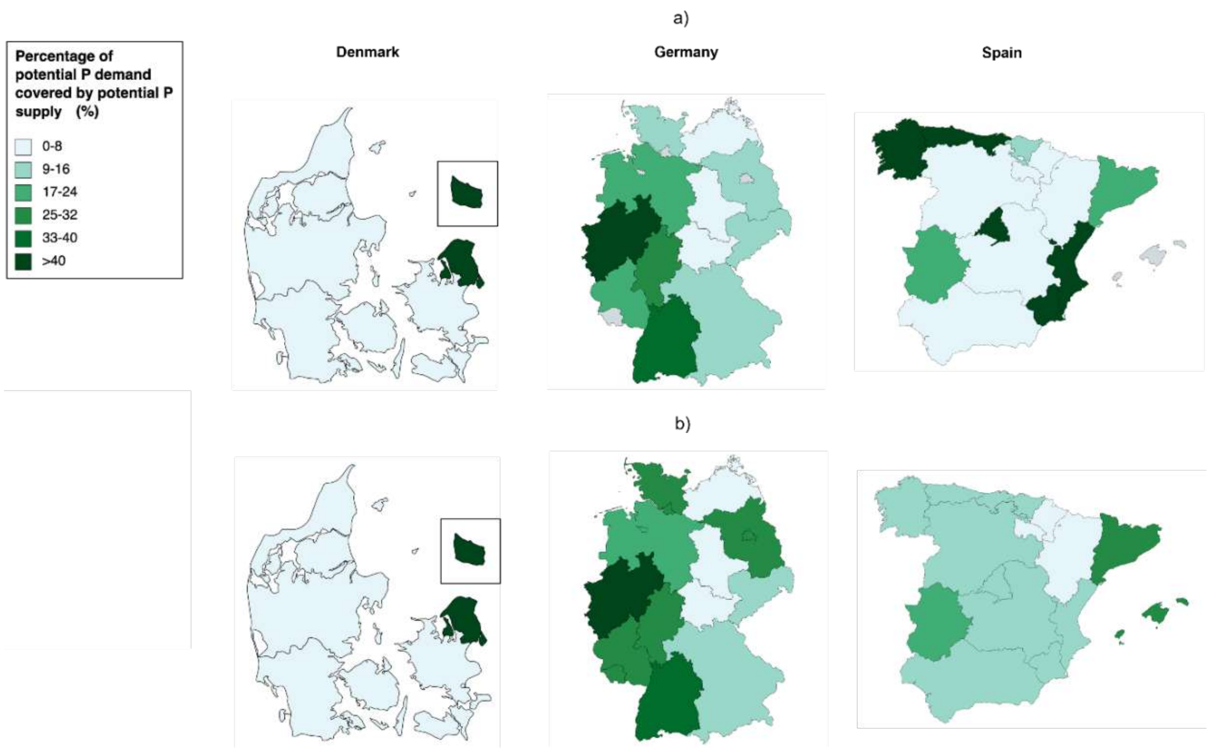

Figure 8. Percentage of regional potential demand covered by potential P supply (a) and percentage of potential P demand covered if regions with a surplus (>100% coverage) combine their potential with neighbouring region (b) in 2018 for Denmark, Germany and Spain. Values of each county's potential supply are associated with the highest supply scenario: Denmark as

vivianite as Ca-P, Germany as ash for Ca-P, and Spain as vivianite as Ca-P with texture and vivianite as Fe-P without texture – see Table A7 for all trade values.

In some cases, the surplus regions were located inside the receiving regions (e.g., Berlin-Brandenburg and Bremen-Lower Saxony, Germany). It was possible to observe more than 10% increase in the potential P demand covered by potential P supply in German regions such as Brandenburg and Schleswig-Holstein, as well as in Andalusia, Spain (see Figure 8b). Although receiving regions such as Castile-Leon (Spain) presented a lower increase, they also have one of the highest potential P demands from all regions in the 3 countries studied (9.94 kg P per capita). Therefore, the market prospect of P recycled products could be analysed in terms of spatial distribution among regions or countries with high potential P supply and low potential P demand and those with opposite conditions.

Spatial distribution analyses, connecting human and animal excreta P supply and crop demand regions, have been done in some countries, demonstrating that logistics are costly even if there are multiple benefits beyond P circularity (e.g. the Netherlands in Lessmann et al., (2023) and Sweden in Metson et al., (2020)). High transport costs associated with the bulky and low nutrient concentrations in sludges and application restrictions due to variable stoichiometric ratios between nutrients can make complete recycling of excreta challenging (Kleinman et al., 2022). In this study, we estimated the potential of recycled P products that have similar characteristics to mineral P fertilisers as an alternative and sustainable P supply for agricultural use. These recycled products could partly cover the P demand for crops with potentially lower logistic costs, but to determine the true potential requires further analysis.

The manufacturing and use of recycled P products evaluated in this manuscript could partially substitute mineral P fertilisers, offering not only benefits in terms of P security via circularity, but also potential environmental benefits. For example, substitution could reduce the risk of Cd accumulation from ~25% to less than 15% (Weissengruber et al., 2018). In countries with high sewage sludge direct reuse in agriculture, the risk of soil pollution with microplastics, estimated approximately in 6 kt per year in the EU at current reuse rates, could be reduced in the same proportion of substitution with recycled P products. Still, quantitatively comparing resource use and pollutions risks between mined and recycled products remains challenging, given that studies use different system boundaries (Manoukian et al., 2023).

3.5 Conclusions

Most P recycling technologies for P from wastewater either do not provide the product information required or currently do not meet the majority of legislative guidelines to be used across multiple EU countries. Only five out of the 22 technologies we reviewed were in full compliance with EU and national legislations in Denmark, Germany, and Spain. We conclude that there is still a need to provide complete information on product characteristics by technology providers, as it is the first step to comparing recycled P products with conventional mineral P fertilisers.

Still, for those technologies that were compliant, we observed that a notable proportion of P demand of selected crops (wheat, barley, and rye, which are produced in around 80% of the total cropland in the selected countries) could be covered by the potential P supply from P recycling technologies best suited for different countries. The most suitable product identified for Denmark was vivianite for calcium phosphate precipitation (0.38 kg P per capita), whereas in Germany ashes for calcium phosphate (0.29 kg P per capita) were the most suitable technology. Lastly, in the case of Spain, vivianite as iron supply was the most suitable product, with 0.05 kg P per capita. These represented 9.1, 21.7 and 10.0 percent (Denmark, Germany and Spain, respectively) of the total potential P demand estimated for wheat, barley and rye production in the studied countries.

Our estimates of P demand, present a useful and systematically compiled figure, but do not represent the full variability of crop requirements on the landscape. For instance, provided that in Denmark and Spain the digested sludge reuse in agriculture is still commonly practiced, there is more potential for sludge-based recycled P products than for ash-based. In Germany, in turn, the potential is higher for ash-based products, and it is expected to increase due to an increase of sludge incineration rates as a consequence of legislation changes in 2017 that restrict sludge reuse in agriculture. Nonetheless, the estimation of the potential P supply was restricted due to a lack of information not only on sludge disposal routes at a regional level but also on the wastewater P inflow reported. Moreover, detailed information on soil properties which affect P fertilisation from a given product can be taken up by crops is necessary to accurately estimate potential substitution of mineral P.

From our estimations of potential supply and demand of recycled P, we evaluated possible trade of P among regions with low agricultural P requirements and regions that did not report crop production information or covered over 100% of their potential P demand. As a result, the percentage of demand covered by recycled P products increased by over 5% in most cases, where trade with neighbouring regions was assessed. Consequently, the analysis of the spatial distribution of potential P supply and potential P demand between regions is crucial to improve the P use efficiency in agriculture, thus reducing mineral P dependency and the environmental impact of directly using untreated sewage sludge in agricultural soil and water pollution.

4. Integrated framework to assess advanced phosphorus recycling as a sustainable alternative to sewage sludge in agricultural soils

4.1 Abstract

Advanced phosphorus (P) recycling from wastewater is critical for improving nutrient circularity and reducing soil pollution associated with the direct application of sewage sludge in agriculture. However, few studies evaluate the long-term environmental and economic trade-offs between recycled P products and raw sewage sludge application. This study compares struvite, vivianite, and dicalcium phosphate (CaP) as P alternatives to sludge to mitigate heavy metal accumulation in Spanish agricultural soils. Using data from 27,835 plots, heavy metal accumulation was simulated over 50- and 100-year fertilisation scenarios. The results indicate that continuous sludge application leads to widespread exceedances of zinc, copper, and cadmium, especially in alkaline soils, whereas substitution with recycled products can substantially reduce these risks. Vivianite balances P recycling and costs, CaP offers the best environmental performance but with higher investment, and struvite suits smaller regions prioritising environmental safety. Economic analysis favours advanced recycling over sludge, especially considering externalities such as soil remediation costs. Despite limitations, our findings emphasise the importance of integrating environmental externalities into economic assessments and the value of advanced P recycling for sustainable soil management.

4.2 Introduction

Phosphorus (P) is an essential element for food production and crucial for multiple industrial applications. In the current context, phosphate rock is the only reliable P source, 85% of which is used to produce mineral P fertilisers for agriculture (Brownlie et al., 2022). However, phosphate rock is finite and unevenly distributed across the globe, thereby increasing the risk of food supply disruptions in highly dependent regions (B. Li et al., 2023). To address this issue, recycling P from waste has been highlighted as an alternative to mineral P (Raniro et al., 2025), where recycling refers to the reintroduction of P contained in waste back into an earlier stage of the cycle (Brownlie et al., 2022). Among several P-containing wastes, sewage sludge, a by-product of urban wastewater treatment plants (WWTPs), has emerged as a promising P source for its relatively high P content (15–25 $g \cdot P\ kg^{-1}$ of dry matter), representing over 80% of the P inflow of WWTPs in case of enhanced P removal (Witek-Krowiak et al., 2022).

Although sewage sludge reuse in agriculture presents a possibility for P recycling, it comes with significant challenges. Besides nutrients, sewage sludge also contains heavy metals (e.g., cadmium, lead, and mercury), organic compounds (e.g., pesticides, per- and polyfluoroalkyl substances (PFAS), pharmaceuticals), pathogens, microplastics, among others, all of which pose risks to human and environmental health (Hudcová et al., 2019; Kanteraki et al., 2022). Moreover, the long-term fate of metals from sludge remains largely uncertain because gradual inputs can accumulate in soils over decades, with persistence and bioavailability strongly modulated by soil pH, organic matter, and redox conditions (Weissengruber et al., 2018). For this reason, the direct application of raw sewage sludge in agriculture, while extensive in some countries (e.g., Spain; (Eurostat, 2022b)), has been progressively restricted in many others (e.g., Germany, Austria and Switzerland; (Egle et al., 2023)).

Advanced technologies have been developed to recover P from sewage sludge while reducing potential contaminants, resulting in environmentally safer fertilisers, more commonly calcium phosphates (e.g., dicalcium phosphate (CaP)), struvite (magnesium ammonium phosphate), or vivianite (iron phosphate) (Amann et al., 2018; Egle et al., 2016; Wijdeveld et al., 2022). These recycled P sources can be used as fertilisers in agriculture, with agronomic performance depending on the product characteristics, environment and management practices (Frick et al., 2025; Hernandez-Mora et al., 2025). Recycled products have been shown to exhibit similar efficiency to mineral P fertilisers and can lower the risk of introducing pollutants (i.e., heavy metals) into agricultural soil compared to the direct application of sewage sludge (Amann et al., 2018). However, these technologies require substantial capital and operational costs and, according to previous economic analyses, remain financially less viable than direct sewage sludge application or mineral P fertilisation (Egle et al., 2016).

Existing economic assessments of P recycling focus solely on direct costs and benefits of implementing a new technology, such as disposal or maintenance cost reductions and revenue from product commercialisation (Egle et al., 2016; Maaß et al., 2014). Although some studies have successfully included externalities of less sustainable processes to offset the costs of implementing innovative technologies, these lack integration of technical environmental impact evaluation to compare advanced P recycling technologies and support decision-making (Molinos-Senante et al., 2011). A truly integrative cost–benefit analysis (CBA) should also consider the environmental externalities of direct sludge application in agriculture, particularly the risk of heavy metal accumulation in soils, which could be mitigated through the adoption of advanced, safer recycled P products (Egle et al., 2023).

To address this gap, we propose a CBA framework that integrates the estimation of the environmental impact of direct sewage sludge application in agricultural soils with a CBA of implementing three different P recycling technologies, accounting for externalities such as heavy metal accumulation in soil. Factors such as sludge application rates and soil characteristics (e.g., pH, P status and heavy metal concentrations) vary among regions (Tóth et al., 2014; Tóth, Hermann, Da Silva, et al., 2016). Therefore, our methodology serves as a model for comparing the environmental and economic performance of P recycling technologies for local agricultural and wastewater treatment infrastructure. By taking Spanish regions as an example, we highlight the importance of incorporating environmental externalities into policy and decision-making considering different regional conditions.

4.3 Materials and methods

The framework is delimited by the pathway from wastewater treatment to agricultural soil application (Figure 9). Sewage sludge generated at wastewater treatment plants can either be directly applied to soil or processed into alternative P fertilisers, with struvite, vivianite and CaP holding particular importance (Hernandez-Mora et al., 2025). We estimated the heavy metal accumulation and economic performance of the three aforementioned recycled P fertilisers and compared them with sludge, which represents the standard practice in Spain.

The framework accounts for direct costs, including energy use, chemical agents, operator labour, transportation, and capital expenditure (CAPEX) associated with the production of the recycled fertilisers. Direct benefits were represented by P savings through avoided mineral fertiliser use, and revenue from potential P surplus sales. Environmental externalities were incorporated in the model by including the cost of soil remediation in contaminated areas and the opportunity cost arising from foregone crop production when sludge application is restricted. Collectively, these elements form the analytical basis for evaluating the integrated environmental and economic performance of P recovery strategies at the regional scale in Spain to promote sustainable soil management.

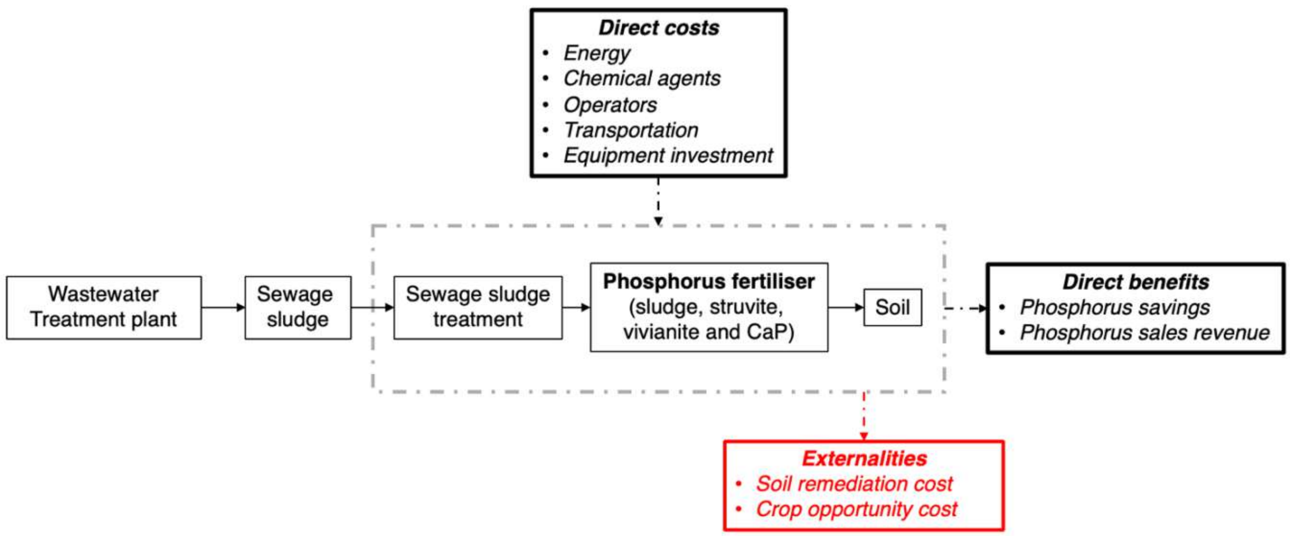

Figure 9. Schematic representation of system boundaries for the use of P from wastewater sewage sludge: current sewage sludge P (baseline scenario) and alternative recycled P (struvite, vivianite, and CaP).

4.3.1 Sewage sludge application and heavy metal content in soil

In a first step we estimated the potential heavy metal accumulation resulting from scenarios involving the application of four P fertilisers in Spanish agriculture: sewage sludge, struvite, vivianite, and CaP, over 50- and 100-year periods. Simulation periods of 50 and 100 years were selected to represent medium- and long-term management horizons commonly used in soil contamination and nutrient balance studies (Merdy et al., 2024; Weissengruber et al., 2018). The 50-year horizon approximates the typical timescale of agricultural soil use under consistent management, whereas the 100-year period enables assessment of potential legacy effects from slow-accumulating heavy metals.

This analysis is based on plot-specific soil characteristics, the composition of sewage sludge and P recovery products, as well as an accumulation model and applicable regulatory thresholds.

4.3.1.1 Sewage sludge application in Spanish agriculture

The use of sewage sludge in agricultural soil is regulated by Spanish Law 1310/1990, which includes guidelines for sewage sludge application and reporting. This information was compiled by the Spanish Ministry of Ecological Transition and Demography in a database from 2018, containing the physicochemical (e.g., heavy metal content, pH, P content) and microbiological characteristics of the sludge used in over 28,000 agricultural plots across all Spanish regions (Ministerio para la Transición Ecológica y el Reto Demográfico, 2021). The database also provides detailed information about the plots such as crop type, coordinates, application area, amount of sludge applied, and WWTP details (code, capacity and location) that supplied the sludge (Table 3). However, only the Spanish regions Andalusia, Castile-La Mancha, Castile-Leon, Catalonia, Extremadura, Galicia, Madrid, Murcia, Rioja and Valencian Community were included in our study, as the nine remaining regions (Asturias, Cantabria, Pais Vasco, Navarra, Aragon, Balearic Islands, Ceuta, Melilla and Canary Islands) did not report the use of sewage sludge in agricultural soil.

Table 3. Data on sewage sludge application across Spanish autonomous communities, detailing number of plots, area in ha (mean of 5.9 ha per plot), mass in t (mean of 5.5 t of sludge per plot), and mean heavy metal concentrations (Cd, Cu, Ni, Pb, Zn, Hg, Cr, P) in mg·kg−1 sludge.

Community	Plots	Area	Mass	Cd	Cu	Ni	Pb	Zn	Hg	Cr	P
Andalusia	11	877.0	253.5	1.0	268.0	19.8	25.4	554.5	0.3	20.2	8,897.2
Castile-La Mancha	10,777	105,903.6	57,773.5	1.5	260.6	35.7	41.9	779.0	5.3	73.6	20,823.6
Castile-Leon	133	1,885.5	12,608.1	1.8	364.4	52.7	100.0	1,367.5	0.7	123.0	13,282.6
Catalonia	13,804	9,962.0	35,706.7	1.3	470.9	61.9	59.2	1,275.0	1.1	110.9	27,631.9
Extremadura	409	20,634.3	24,939.4	1.2	206.6	24.0	43.3	511.2	0.5	38.2	10,736.7
Galicia	3	69.7	573.7	0.3	13.0	10.0	1.3	63.3	0.0	48.7	207.0
Madrid	85	941.3	1,419.0	2.1	361.2	30.8	44.7	541.1	0.0	101.1	24,505.7
Murcia	1,452	19,966.1	12,595.7	1.6	239.4	15.3	33.8	660.5	0.3	35.9	19,213.6
Rioja	1,154	3,479.8	6,773.7	1.9	239.9	44.6	45.2	1,111.6	0.4	92.5	15,847.6
Valencian Community	7	17.0	45.0	2.0	401.0	21.0	44.0	1,616.0	1.0	33.0	10,469.9

National average	1.4	362.9	47.8	50.5	1,030.8	2.6	90.7	23,727.2

4.3.1.2 Soil characteristics of plots

Soil properties of each plot were obtained using the Zonal Statistics Tool of QGIS to analyse the raster files from the European Soil Data Centre database and extract the values of cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb), zinc (Zn), mercury (Hg), chromium (Cr) and pH at the exact coordinates (Ballabio et al., 2019; Tóth, Hermann, Szatmári, et al., 2016). In the case of exact coordinate points, the Zonal Statistics tool uses each plot coordinate as a 'zone' and samples the coincident cell value from the raster files (Cd, Cu, Ni, Pb, Zn, Hg, Cr and pH), extracting the exact value. Some plots (n = 292) presented a surface area of zero, likely owing to rounding or loss of decimal precision during file-type conversion, thus these records were excluded from the analysis. Similarly, plots with a reported sludge application mass of zero (n = 604) were removed, resulting in a final dataset comprising 27,835 plots. In some plots, the points were rendered outside the raster maps (28 points for Zn and 216 for pH), and these values were replaced with the overall average (Figure 10).

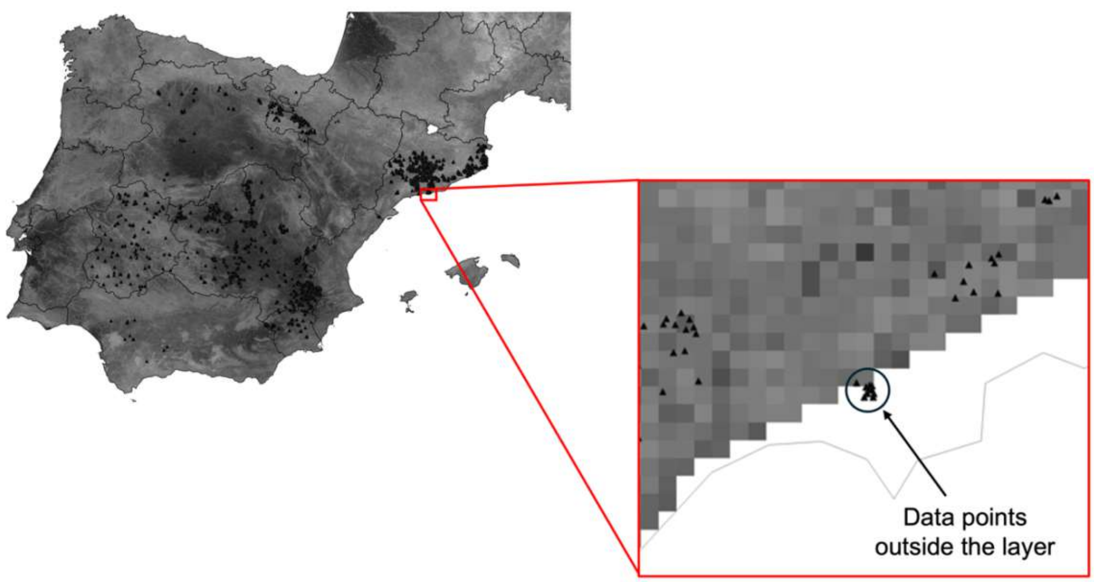

Figure 10. Sample group of agricultural plots (black triangles) where sewage sludge is applied and were rendered outside the Zn raster layer (n = 28).

4.3.1.3 Scenarios and system boundaries

Heavy metal pollution resulting from sludge-based P fertilisation originates at the WWTP, and its environmental impact may extend to water bodies (Narayanan et al., 2025; Sharafi & Salehi, 2025; Su et al., 2025). However, the transfer of heavy metals from soil to water bodies primarily occurs through leaching and runoff, which are highly variable flows that depend on location-specific factors, such as soil characteristics, slope, precipitation, and proximity to ground or surface water (C. Liu et al., 2025). Consequently, our analysis focuses on the long-term heavy metal accumulation in soil and its economic implications for regional Spanish agriculture.

Before sewage sludge can be used in agriculture (Table 4), it is in most cases treated to reduce pathogens and odours (*RD 1051/2022*, 2022; *Sewage Sludge Directive (86/278/EEC)*, 1986). This includes stabilisation to reduce organic matter and odours, usually through anaerobic or aerobic digestion (Yoshida et al., 2015). The sludge is then dewatered to lower its moisture content, making it easier to handle and transport for application to agricultural soil (Yoshida et al., 2015).

Table 4. Comparative summary of sewage sludge and alternative phosphorus recycling technologies (Egle et al., 2015; Santos et al., 2021; F. Zhu et al., 2023).

Phosphorus Fertilisers	Source Stream	Recovery Efficiency	Advantages	Disadvantages
Sludge (baseline scenario)	-	-	No investment required	High heavy metal content
Struvite	Sludge liquid phase	>95 % (~30% P inflow)	High agronomic value fertiliser	pH sensitive, high operational costs
Vivianite	Sludge	>80 % of vivianite-bound P (~60–65 % P inflow)	Low operational costs	High chemical consumption (additional iron salts)
CaP	Sludge ashes	>95 % (~80% P inflow)	Removes additional pollutants (e.g., microplastics)	High capital investment

In contrast to direct application, P can be recovered from sludge using different chemical and physical processes. Struvite, vivianite, and CaP can be produced depending on the treatment method, chemical dosing, and infrastructure in place at WWTPs (Table 4) (F. Zhu et al., 2023). First, after dewatering the sludge, the liquid phase (or press water) is recirculated back to the treatment step and contains up to 20–30% of P inflow to WWTP. This press water can be precipitated as struvite by dosing a magnesium salt (typically magnesium chloride) into a crystallisation reactor (C. Fang et al., 2016; Z. G. Liu et al., 2021). Alternatively, P in sludge binds with iron to form vivianite under anaerobic conditions when sufficient iron is present in the sludge, by dosing iron salts (typically iron chloride). Through magnetic separation, 60–65% of the total P in the WWTP inflow can be recovered (Prot et al., 2019; Wijdeveld et al., 2022). Yet another possibility is the incineration of the s(Law & Pagilla, 2021)ed sludge (Law & Pagilla, 2021), producing sludge ashes. These ashes contain up to 80% of the P inflow to the WWTP and can be chemically treated (acid leaching) to recover P with over 95% efficiency (H. Liu et al., 2021).

The varying chemical characteristics of the four P fertilisers studied provide advantages or disadvantages depending on soil conditions. In the context of heavy metal accumulation, their heavy metal content is a crucial factor (Table 5). In our framework, each scenario refers to the long-term application of a specific fertiliser product, where the baseline scenario corresponds to the current practice of sewage sludge use, and the alternatives correspond to struvite, vivianite, and CaP.

Table 5. Phosphorus and heavy metal content (mg kg–1 fertiliser) of studied fertilisers (Serrano-Gomez et al., 2023; Weissengruber et al., 2018; Wijdeveld et al., 2022).

	P	Cd	Cu	Ni	Pb	Zn	Hg	Cr
	$g \cdot kg^{-1}$				$mg \cdot kg^{-1}$			
Sludge—national average (baseline scenario)	23.7	1.4	363	47.8	50.5	1031	2.6	90.7
Struvite	169	0.4	19	5	6	260	0.1	14
Vivianite	104	0.4	130	42	15	380	0.2	120
CaP	169	0.1	5	2.5	3.6	15	0.1	1.7

4.3.1.4 Heavy metal accumulation and soil limits

In Spain, the permissible concentrations of heavy metals in agricultural soils are regulated by the Spanish Soil Nutrition Decree 1051/2022, which distinguishes between acidic (pH < 7) and alkaline (pH ≥ 7) soils (Table 6; (*RD 1051/2022*, 2022)). Accordingly, the heavy metal limits varied among plots depending on soil pH.

Table 6. Heavy metal concentration limits for agricultural soils with different pH according to Spanish legislation (*RD 1051/2022*, 2022).

	Limit Values (mg·kg^{-1} Soil)	
	pH < 7	**pH ≥ 7**
Cd	20	40
Cu	1000	1750
Ni	300	400
Pb	750	1200
Zn	2500	4000
Hg	16	25
Cr	1000	1500

The heavy metal accumulation was estimated as the annual change in soil concentration for each plot (i) by yearly (t) heavy metal input of the respective applied P source (Equation 4) as follows:

$$c_{soil_{i,t+1}} = c_{soil_{i,t}} + \frac{m_{input_{i,t}}}{M_{soil}} \text{ (Equation 4),}$$

where the mass input of a heavy metal ($m_{input_{i,t}}$) spread across a certain soil mass (M_{soil}) was added to the actual concentration of that heavy metal for year t ($c_{soil_{i,t}}$) to obtain the accumulated concentration for the following year $t+1$ ($c_{soil_{i,t+1}}$). in the top 0.25 m layer of soil, with a bulk density of 1.2 t soil·m^{-3} resulting in a soil mass of 3000 t soil·ha^{-1} (Rodríguez Martín et al., 2016).

Soil leaching and plant uptake flows were not included in the accumulation model, as their contribution to element output has been shown to significantly decline with increasing soil pH and decreasing precipitation (Weissengruber et al., 2018). This is consistent with national conditions in Spain, except in some of its regions, namely Galicia, Asturias, Cantabria and Pais Vasco (Dankers & Hiederer, 2008; Tóth, Hermann, Szatmári, et al., 2016). For this reason, and to ensure consistent conditions across all fertilisation scenarios, we applied a mass balance that focuses on differences in heavy metal inputs between sludge, struvite, vivianite, and CaP.

4.3.2 Cost-benefit analysis

The cost–benefit analysis evaluates the economic viability of substituting sewage sludge with advanced P recycling technologies by incorporating direct costs and benefits, environmental externalities, and CAPEX. It also includes the calculation of the net present value (NPV) and the benefit–cost (B/C) ratio to support policy-relevant comparisons between treatments.

4.3.2.1 Capital cost of phosphorus recycling technologies

In our cost–benefit framework, the CAPEX for P recycling was annualised over 25-year cycles with full reinvestment at the end of each cycle. For the 50- and 100-year horizons considered, this assumption implies one and three full reinvestments, respectively. In the baseline scenario of sewage sludge application, existing wastewater treatment infrastructure was assumed to be already available during the first 25 years, so CAPEX was set to zero for this initial period. After year 25, reinvestments of the baseline infrastructure were also included, consistent with the alternative scenarios.

For the recycling technologies, there are references regarding the CAPEX required for a specific WWTP size (see Table A10) (Egle et al., 2016; Ehsan, 2021; Kelessidis & Stasinakis, 2012; Uzkurt Kaljunen et al., 2022). Through the WWTP code, we obtained the capacity of all the WWTPs that provided sludge for agricultural use in the registered plots. However, most of the WWTP sizes were different from the reference, and thus the CAPEX was adjusted to the actual scale of the WWTP through a scale–CAPEX relationship (Equation 5; (Tsagkari et al., 2016)):

$CAPEX = CAPEX_r \cdot \left(\frac{S_r}{S}\right)^{0.6}$ (Equation 5),

where $CAPEX_r$ and S_r correspond to the CAPEX required for a reference WWTP size, while $CAPEX$ and S represent the CAPEX required for the actual WWTP size that provided the sludge for agricultural use.

In the case of the alternative recycling technologies, total CAPEX includes both the baseline wastewater treatment infrastructure and the additional technology required for P recovery. This ensures comparability across scenarios by assuming that all WWTPs must meet the same baseline requirements.

Most of the WWTPs supplying sludge for agricultural use applied only part of their total sludge production. In contrast, implementing P recycling technologies requires investment at the scale of the full WWTP, since recovery units treat the entire sludge stream rather than the marginal share applied to agricultural plots. As a result, the recovered P generally exceeded the regional agricultural demand, creating a surplus that could be commercialised. Therefore, for the recycled P scenarios, we included the sales revenue of P surplus (if available) within the benefits described in Section 2.2.2.

4.3.2.2 Direct costs and benefits

The direct benefits (B_t) were composed by the savings from mineral P replacement and, in the case of recycled P products, sales revenue from P surplus (Equation 6). To account for these benefits, we assumed a price of 2000 EUR t^{-1}·P, based on the commercial price of triple superphosphate, a commonly used mineral P fertiliser (Egle et al., 2016).

$$B_t = (p_P \cdot m_{P_{used}}) + \left(p_P \cdot m_{P_{surplus}}\right) \text{ (Equation 6)},$$

where $m_{P_{used}}$ is the mass of P applied to soils, p_P price of P, and $m_{P_{surplus}}$ the surplus recycled P available for sale.

In contrast, direct costs (C_t) included the costs of treatment ($C_{treatment}$) (e.g. energy, chemicals, waste disposal), transportation ($C_{transport}$) and annualised CAPEX ($CAPEX$) (Egle et al., 2016; Uzkurt Kaljunen et al., 2022) (Table A10):

$$C_t = (C_{treatment} \cdot m_{P_{used}}) + (C_{transport} \cdot d \cdot m_{P_{used}}) + CAPEX \text{ (Equation 7)},$$

where $m_{P_{used}}$ denotes the mass of P applied to soils.

4.3.2.3 Externalities

Externalities in the CBA were represented by the costs of the soil pollution with heavy metals. Specifically, we accounted for soil remediation costs (C_{rem_i}) when legal thresholds were exceeded, and opportunity costs (C_{op_i}) from lost crop production in contaminated plots, that is, the cost of not being able to produce agricultural products due to the exceedance of heavy metal legal limits (Schaub et al., 2023). We incorporated remediation costs at the year t when heavy metal limits were first exceeded in each plot, while the opportunity cost was applied annually starting from the same year of exceedance.

Remediation costs vary (10-1000 EUR t^{-1} soil) depending on the technique (Ehsan, 2021). We considered the *in-situ* phytoremediation technique, a commonly used technique for heavy metal contamination remediation (L. Liu et al., 2018), costing around 100 EUR t^{-1} soil (Ehsan, 2021). The crop prices were obtained from the Spanish national account (Ministry of Agriculture, 20 C.E.) (Table A9).

$C_{rem_i} = C_{rem} \cdot A_i$ (Equation 8),

$C_{op_i} = p_{crop} \cdot A_i$ (Equation 9),

where p_{crop} is the price of the specific crop grown at each plot per hectare and A_i the plot area.

4.3.2.4 *Economic performance and sensitivity analysis*

To evaluate the economic case, we used two commonly applied indicators of efficiency for each phosphorus fertilisation scenario in region *j*: Net Present Value (NPV) in million euros (EUR) and the benefit–cost (B/C) ratio.

$NPV_j = \sum_{t=0}^{T-1} \frac{B_{j,t} - (C_{j,t} + CAPEX_{j,t} + C_{op,j,t} + C_{rem,j,t})}{(1+r)^t}$ (Equation 10),

$B/C_j = \frac{\sum_{t=0}^{T-1} \frac{B_{j,t}}{(1+r)^t}}{\sum_{t=0}^{T-1} \frac{C_{j,t} + CAPEX_{j,t} + C_{op,j,t} + C_{rem,j,t}}{(1+r)^t}}$ (Equation 11),

where $B_{j,t}$ and $C_{j,t}$ are the direct benefits and costs in year t, $CAPEX_{j,t}$ is the annualised capital cost of infrastructure, $C_{op,j,t}$ represents the opportunity cost, which was zero until the year of exceedance and positive thereafter, and $C_{rem,j,t}$ denotes the remediation cost, which was zero in all years except the specific year of exceedance when it is applied once.

The NPV quantifies the difference between the present value of benefits and the present value of all related costs over time. Therefore, a positive NPV suggests that the advantages (e.g., avoided remediation costs) outweigh the investments associated with(Bianchini & Rossi, 2020)liser (Bianchini & Rossi, 2020). The B/C ratio reflects the relative efficiency of the investment, where values greater than one indicate that benefits exceed costs, and values below one suggest the opposite (Bianchini & Rossi, 2020). In our context, interpreting these indicators enables us to assess phosphorus recycling not as a purely commercial return but as an environmental service, where slightly negative NPVs can still be worthwhile if they deliver lasting benefits to soil health, reduced pollution, and agricultural sustainability.

For projects where there are potential long-term impacts on society and the environment, the European Commission recommends to its Member States and Cohesion countries considering 3–5% discount rate (Sartori et al., 2015). To provide a conservative scenario, we considered 5% discount rate. In addition, to assess the robustness of the economic evaluation, we conducted a sensitivity analysis by varying the discount rate (r) from 3% to 7% for the NPV and B/C ratio for each P fertilisation scenario.

In terms of implementation, the sludge scenario assumed no capital expenditure during the first 25 years, since existing wastewater treatment infrastructure was already in place. From year 25 onwards, the baseline annuity was applied. For struvite, vivianite, and CaP, technology-specific annuities were included from the beginning, and both technology and baseline annuities were applied from year 25 onwards. Opportunity costs were accounted for annually starting from year 25, while remediation costs were charged only once at the end of the planning horizon.

4.3.3 Limitations of the methodology

There are limitations inherent to the methodology that could introduce inaccuracies in estimates of heavy metal accumulation and economic performance. Uncertainty in long-term soil heavy metal behaviour was not quantified explicitly. In particular, the exclusion of leaching and plant uptake introduces uncertainty regarding the exact timing of threshold exceedance. However, such processes depend on location-specific unpredictable factors such as rainfall, which can vary substantially over time. Moreover, we simplified the estimation of non-financial costs of heavy metal accumulation, excluding the costs of health risks from contaminated crops, as well as further environmental impacts such as water pollution (Amann et al., 2018). Our framework is limited to the economic effects of heavy metals in agricultural soils resulting from sludge input, while further pollutants present in sludge such as PFAS, pharmaceuticals, pathogens and microplastics could pose further environmental and health risks (Arvaniti et al., 2024; Xue et al., 2025). Conversely, the framework does not account for potential agronomic benefits of using sludge in agriculture, such as increased soil organic carbon and inputs of additional nutrients

(Kanteraki et al., 2022). While the extent of these inaccuracies cannot be precisely quantified, they likely affect both costs and benefits, and their omission should be considered when interpreting the results, especially in borderline cases.

4.4 Results and discussion

4.4.1 Heavy metal accumulation in agricultural soils

Most plots exhibited alkaline or neutral soils, particularly in Catalonia, Castile-Leon, Murcia and Rioja (13,316, 10,344, 1445 and 1151 plots, respectively) (Figure A2). Acidic soils were mostly found in plots from Catalonia (n = 488), Castile-La Mancha (n = 433) and Extremadura (n = 315). In contrast, Andalusia, Rioja, Valencian Community and Murcia showed minimal or no use of sewage sludge on acidic soils, while in Galicia all plots had acidic soils. This distribution reflects regional variability in soil properties and agricultural practices, both of which influence sludge application and heavy metal accumulation. Increased soil pH generally reduces dissolved Cd, Cu, Ni, Pb, and Zn via adsorption and carbonate/hydroxide formation, lowering mobility and plant availability, and thus, favouring in-soil stock build-up under repeated inputs. This immobilisation leads to lower leaching risks in alkaline soils, while at the same time, contributes to the accumulation of heavy metals. Conversely, in acidic soils, higher heavy metal mobility leads to more losses and uptake, leading to lower accumulation in the long term (Meng et al., 2023).

4.4.1.1 Soil after direct sewage sludge application

Our results highlight the complex interplay between long-term sludge application, heavy metal accumulation, and soil pH, emphasising the need for region-specific management strategies (Figure 11). After 50 years of sludge input, most plots still remained within regulatory limits, but approximately 9% exceeded at least one heavy metal threshold (n = 2447), with a significant portion (n = 251) surpassing the limits for multiple metals. Soil pH strongly influences metal retention and mobility, with acidic soils typically promoting higher metal solubility and leaching (Kicińska et al., 2022; Xu et al., 2022), which can reduce accumulation and thus lower exceedance rates under comparable input conditions (Arvaniti et al., 2024; Kicińska et al., 2022). However, in certain regions like Rioja (63%), Catalonia (48%), and Castile-Leon (37%), exceedance rates in acidic soils were disproportionately high. This pattern likely reflects the overriding effect of elevated heavy metal concentrations in the sludge (Table 3) applied in these areas, which were large enough to offset the mitigating influence of low pH on metal accumulation.

As such, heavy metal accumulation results from the interplay between soil pH and sludge characteristics and cannot be attributed to either factor alone.

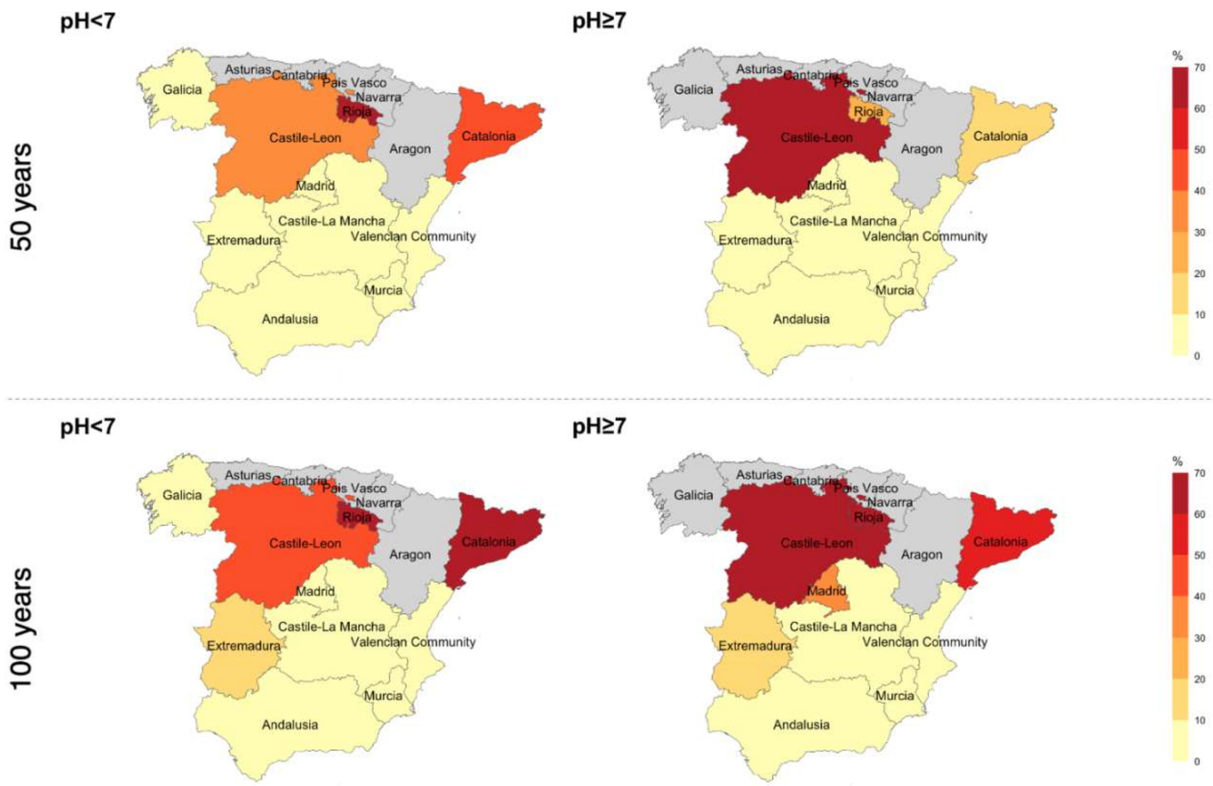

Figure 11. Percentage of total plots that, after applying sewage sludge for 50 and 100 years, exceeded at least one heavy metal concentration limit with pH < 7 and pH ≥ 7. Grey regions (namely Asturias, Cantabria, Pais Vasco, Navarra, Aragon, Balearic Islands, Ceuta, Melilla and Canary Islands) did not register sludge application.

We observed long-term shifts in heavy metal accumulation due to sustained sludge application (Figure 12), particularly emphasising the predominance of Zn and Cu exceedances after 50 years. These two elements are essential micronutrients for plant growth, yet their accumulation beyond legal thresholds raises concerns. At excessive concentrations, Zn can inhibit root development and microbial activity (Meng et al., 2023), while Cu can generate oxidative stress in plants, inactivate key enzymes for plant metabolism, and reduce soil biodiversity and consequently enzymatic activity in soil (Duan et al., 2022). Although these metals pose relatively lower toxicity risks compared to other heavy metals (Cd, Hg and Pb), their continued build-up suggests that prolonged sludge application could gradually transition from nutrient enrichment to toxicity concerns. In addition, excessive Zn accumulation in soils can further reduce P availability by promoting the formation of insoluble Zn-phosphate complexes, which limit P solubility and uptake by plants, potentially exacerbating P deficiency and reducing P fertilisation efficiency in agricultural systems (Ding et al., 2021).

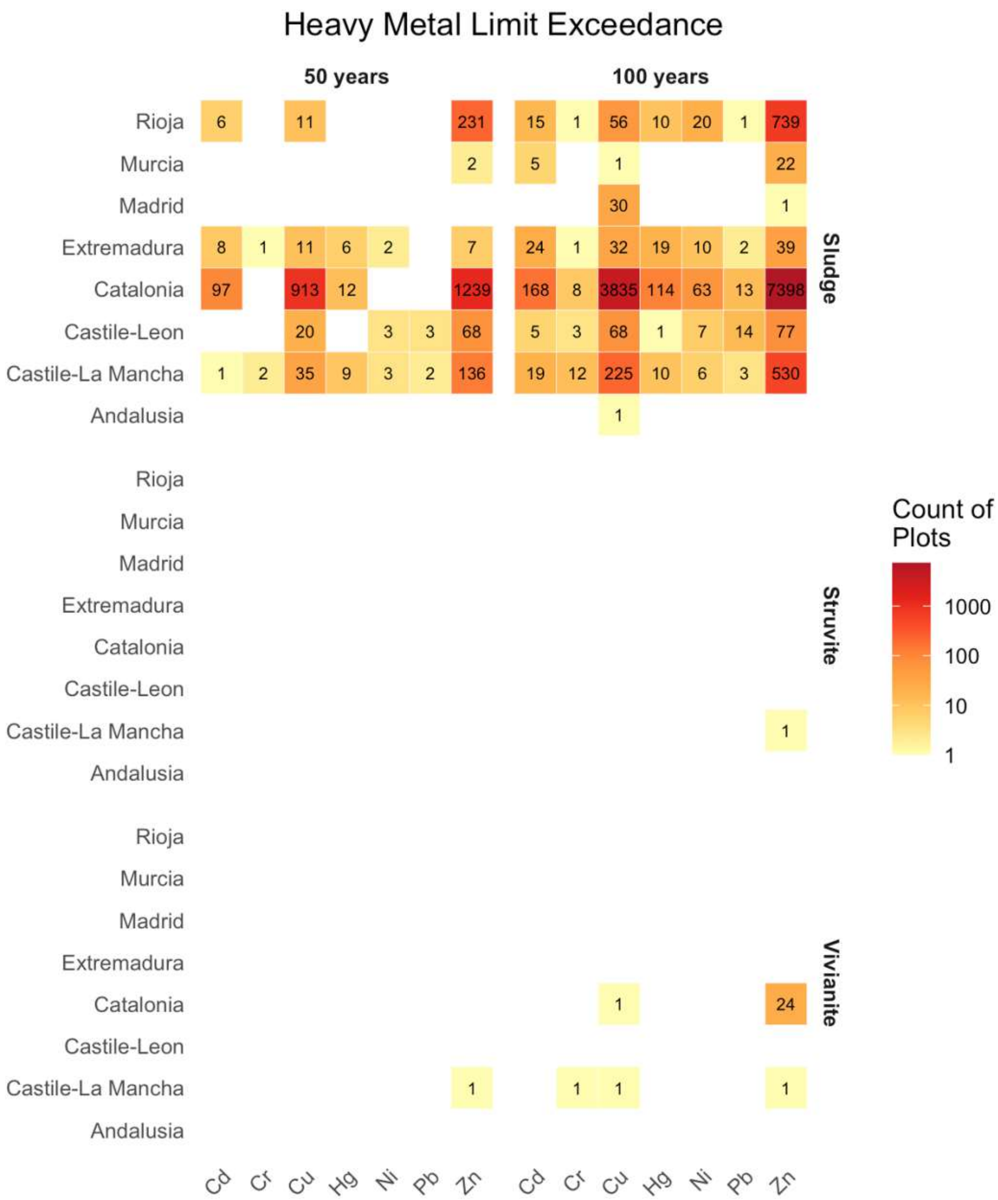

Figure 12. Distribution of heavy metal concentration limits exceeded after 50 and 100 years of P fertilisation by region, heavy metal and P source. The CaP scenario did not exceed any heavy metal limits in both time frames. Galicia and Valencian Community did not present any plots exceeding concentration limits in any scenario and time frame.

A more alarming trend emerges when considering the implications of sludge application over a century. In Murcia's alkaline soils, Zn is currently the sole heavy metal exceeding limits in 100% of affected plots after 50 years. However, projections for 100 years of direct sewage sludge application reveal that Cd and Cu would also surpass regulatory limits, accounting for 18% and 4% of total plots exceeding limits, respectively. Given that Cd is highly toxic, bioaccumulative, and classified as a carcinogen, its increasing concentration in soil raises serious environmental and health concerns, as Cd can enter the food chain through plant uptake and groundwater via leaching (Ankush et al., 2024). Similar trends have been observed in previous studies, where carcinogenic heavy metal concentration in agricultural soil via sludge and mineral P could reach a critical concentration where leaching and plant uptake could lead to health issues and environmental deterioration (Briffa et al., 2020).

Our findings underscore the need for long-term sludge management strategies, particularly in alkaline regions where metals are more likely to accumulate than leach. While Zn and Cu are essential at trace levels, their gradual accumulation and the eventual increase in carcinogenic metals such as Cd necessitate periodic soil monitoring, stricter regulations, and possible amendments to mitigate risks. Without intervention, continued sludge application could lead to soil degradation, reduced agricultural productivity, and heightened risks of metal transfer to water sources, challenging the sustainability of sludge use in agriculture over extended periods.

4.4.1.2 Soil after recycled phosphorus application

According to our model, the alternative application of struvite or CaP would lead to no heavy metal limit exceedance in any plot over the 50-year horizon (Figure 12). However, the use of vivianite would lead to Zn exceedance in one plot located in Castile-La Mancha. Zinc accumulation associated with vivianite application is due to sewage-sludge-derived vivianite, typically containing trace metals inherited from the sludge matrix. Consequently, even though vivianite is primarily composed of Fe and P, its Zn content can be higher than that of other recycled fertilisers and contribute to long-term soil accumulation when applied repeatedly.

Even after 100 years, no plots would exceed the limits by applying CaP, but in the case of struvite and vivianite, the number of plots would increase to one (in Castile-La Mancha) and 28 (three in Castile-La Mancha and 25 in Catalonia), respectively.

A considerably lower number of plots transgressed heavy metal limits when recycled P sources were compared to sludge. These results were expected, as the heavy metal concentration per kilogram of P present in the recycled products is much lower than in the sewage sludge used in Spanish cropland. Similar results were observed in previous reports (Egle et al., 2023; Weissengruber et al., 2018), although in one study, the long-term heavy metal accumulation from direct sludge application was lower, mainly due to the difference in soil pH values (Weissengruber et al., 2018).

Advanced P recycling also presents potential advantages in terms of lower input of contaminants, including PFAS and pharmaceuticals. While such pollutants are commonly found in sewage sludge, processes like crystallisation or thermochemical conversion involved in struvite and CaP production reduce their presence, improving the environmental safety of recycled fertilisers (Vogel et al., 2023). Furthermore, vivianite appears especially promising for the Spanish context due to its compatibility with flooded paddy soils used in rice production and its role in iron-mediated P availability (Saracanlao et al., 2024). However, its fertiliser value is lower compared to other P products due to its reduced solubility under non-flooded conditions, limiting its application to specific crops or soil types (Hernandez-Mora et al., 2025).

Regional variations in heavy metal accumulation observed are primarily driven by differences in sludge characteristics and amounts applied, and soil pH, influencing both metal mobility and accumulation trends. Regions with alkaline soils, such as Murcia and Rioja, demonstrated higher long-term heavy metal exceedances, particularly for Zn, Cu, and Cd, resulting from the continuous application of sewage sludge. Conversely, recycled P products such as struvite, vivianite, and CaP significantly reduce these exceedances due to their inherently lower metal contents. Consequently, regions applying sludge with higher pollutant loads or possessing alkaline soils incur greater environmental externality costs associated with soil remediation and lost agricultural productivity. Ultimately, these regional differences in heavy metal accumulation directly shape the environmental externalities and influence the economic performance of recycled P fertilisers compared to direct sewage sludge application.

4.4.2 Economic performance of alternative phosphorus recycling in comparison to direct sludge application

While almost all NPVs across regions remained negative, the results indicate that some recycled P products performed relatively better than direct sewage sludge application, pointing to both economic and environmental advantages of P recycling technologies (Table 7). This relative advantage highlights the role of recycled P not necessarily as a commercially profitable venture, but as a public environmental service of reducing the pollutant burden of sewage sludge application while returning valuable nutrients to agriculture in a safer way.

Table 7. Net present value (NPV) in million EUR and benefit–cost (B/C) ratio of all P fertilisers after 50 and 100 years of application by region and treatment.

		Net Present Value (NPV) [Million EUR]		**Benefit–Cost Ratio (B/C)**	
		50 years	100 years	50 years	100 years
Andalusia	*Sludge*	–1.12	–0.89	0.13	0.13
	Struvite	–2.04	–1.68	0.11	0.11
	Vivianite	–1.31	–1.14	0.25	0.24
	CaP	–3.60	–3.29	0.12	0.10
Castile-La Mancha	*Sludge*	–344.54	–268.69	0.11	0.11
	Struvite	–530.38	–437.57	0.13	0.12
	Vivianite	–334.21	–293.67	0.29	0.27
	CaP	–941.15	–863.19	0.13	0.12
Castile-Leon	*Sludge*	–67.91	–33.15	0.05	0.08
	Struvite	–29.63	–24.20	0.11	0.11
	Vivianite	–13.46	–11.81	0.29	0.27
	CaP	–41.98	–37.91	0.12	0.11
Catalonia	*Sludge*	–120.51	–88.54	0.24	0.25
	Struvite	–205.75	–168.37	0.32	0.31
	Vivianite	50.29	31.89	1.35	1.26
	CaP	–170.30	–169.39	0.56	0.50
Extremadura	*Sludge*	–111.04	–84.22	0.07	0.08
	Struvite	–156.19	–129.20	0.06	0.06
	Vivianite	–126.45	–109.43	0.14	0.13
	CaP	–308.54	–281.74	0.07	0.06
Galicia	*Sludge*	–1.60	–1.26	0.00	0.00
	Struvite	–1.34	–1.12	0.00	0.00
	Vivianite	–1.58	–1.35	0.00	0.00
	CaP	–3.35	–3.06	0.00	0.00
Madrid	*Sludge*	–10.33	–8.16	0.11	0.11
	Struvite	–18.39	–15.22	0.13	0.13
	Vivianite	–12.07	–10.65	0.32	0.29
	CaP	–34.02	–31.38	0.15	0.13
Murcia	*Sludge*	–63.33	–49.33	0.14	0.14
	Struvite	–114.43	–94.48	0.16	0.16
	Vivianite	–59.39	–53.36	0.43	0.40

	CaP	–192.80	–178.60	0.20	0.18
Rioja	*Sludge*	–60.13	–45.79	0.06	0.06
	Struvite	–84.10	–69.77	0.08	0.07
	Vivianite	–72.88	–63.10	0.15	0.14
	CaP	–175.88	–160.96	0.07	0.06
Valencian Community	*Sludge*	–0.45	–0.35	0.04	0.04
	Struvite	–0.66	–0.55	0.06	0.06
	Vivianite	–0.64	–0.56	0.12	0.11
	CaP	–1.49	–1.37	0.06	0.05

In Catalonia, vivianite achieves a positive NPV in both the 50- and 100-year horizons. This benefit stems from vivianite's high P recovery efficiency, which can capture up to 60% of the influent P in WWTPs and provide high P surplus (Lu et al., 2022), while reducing the externalities associated with heavy metal accumulation. In most other regions, however, vivianite and CaP still reduce environmental costs relative to sludge, but do not reach positive NPVs, underlining the strong influence of regional conditions such as soil pH, pollutant loads, and sludge application rates.

In comparison, struvite demonstrates environmental benefits due to its low heavy metal content, making it a safer option in terms of soil contamination. However, its low P recovery efficiency (20–30% of influent P from WWTP) and relatively high treatment costs limit its cost-efficiency per unit of recycled P when compared to vivianite and CaP. Despite this, we observe an exception in smaller-scale regions such as Galicia and Rioja, where the NPV of struvite was slightly higher than that of CaP. This is attributable to a smaller scale of WWTPs that supply P demand. In such cases, the higher specific cost of struvite (EUR t^{-1}·P recovered) became less penalising due to the smaller absolute volumes involved. These findings imply that struvite may be a favourable option for smaller regions prioritising environmental safety over cost-effectiveness.

In contrast, CaP, which has the highest P recovery efficiency (up to 80% of influent P from WWTP), showed superior environmental performance due to low heavy metal content, eliminating indirect costs associated with soil remediation and cost of opportunity. Nevertheless, its economic appeal is strongly reduced by the high CAPEX, which dominated the cost structure over both 50- and 100-year horizons. Interestingly, while CaP had the lowest NPV in several regions over a 50-year period, its benefit–cost ratio remained relatively stable. This discrepancy reflects how the

B/C ratio captures proportional efficiency rather than cumulative profit. Since CAPEX in the model was not included as a one-time investment but amortised annually, its impact was spread more evenly over time, which explains why the B/C ratio was higher than that of struvite despite lower NPVs. Similar observations were made in prior assessments of P recovery technologies across the EU, which found that high-CAPEX processes, such as CaP, may require regional subsidies to ensure long-term economic viability (Egle et al., 2016). These results confirm that CaP can deliver long-term sustainability advantages if policy mechanisms such as infrastructure subsidies or pollution-based levies are implemented to compensate for the high initial investments.

An overarching trend across all recycled P products is that economic performance improves only marginally over time. With recycled P, B/C ratios remained relatively stable between the 50- and 100-year marks, reflecting how annualised CAPEX dominates the cost structure and how avoided externalities accumulate slowly. This contrasts with direct sludge application, where remediation and opportunity costs grew substantially in the long term. These results indicate that while advanced P recycling reduces environmental risks, its economic viability depends strongly on regional conditions and policy support to overcome the cost barrier.

The sensitivity analysis reveals distinct patterns in how discount rates influence the economic performance of the studied P fertilisers. In the case of Catalonia for an application period of 50 years, the estimated NPV becomes less negative with higher discount rates for struvite and sludge (Figure 13). This is a trend observed in NPV across regions mainly due to less weight in future costs compared to present costs at higher discount rates (see Equation 8 and 9). In contrast, an exception is observed for vivianite in Catalonia, where begins with a positive NPV at lower discount rates, and diminishing with higher rates as future benefits are devalued more steeply. Such trend is comparable to typical NPV behaviour for increasing discount rates (Gollier, 2010). Meanwhile, the B/C ratio remains unchanged for struvite and vivianite, indicating a proportional evolution of benefits and costs that neutralises the impact of the discount factor. In the case of sludge, the B/C ratio improves with increasing discount rates, likely due to the delayed emergence of externalities such as soil remediation. Conversely, the high CAPEX of CaP implementation is penalised in heavily in short-term periods thereby decreasing B/C ratio with increasing discount rates. This underlines how temporal cost structures and externalities affect the sensitivity of investment metrics in P recycling pathways.

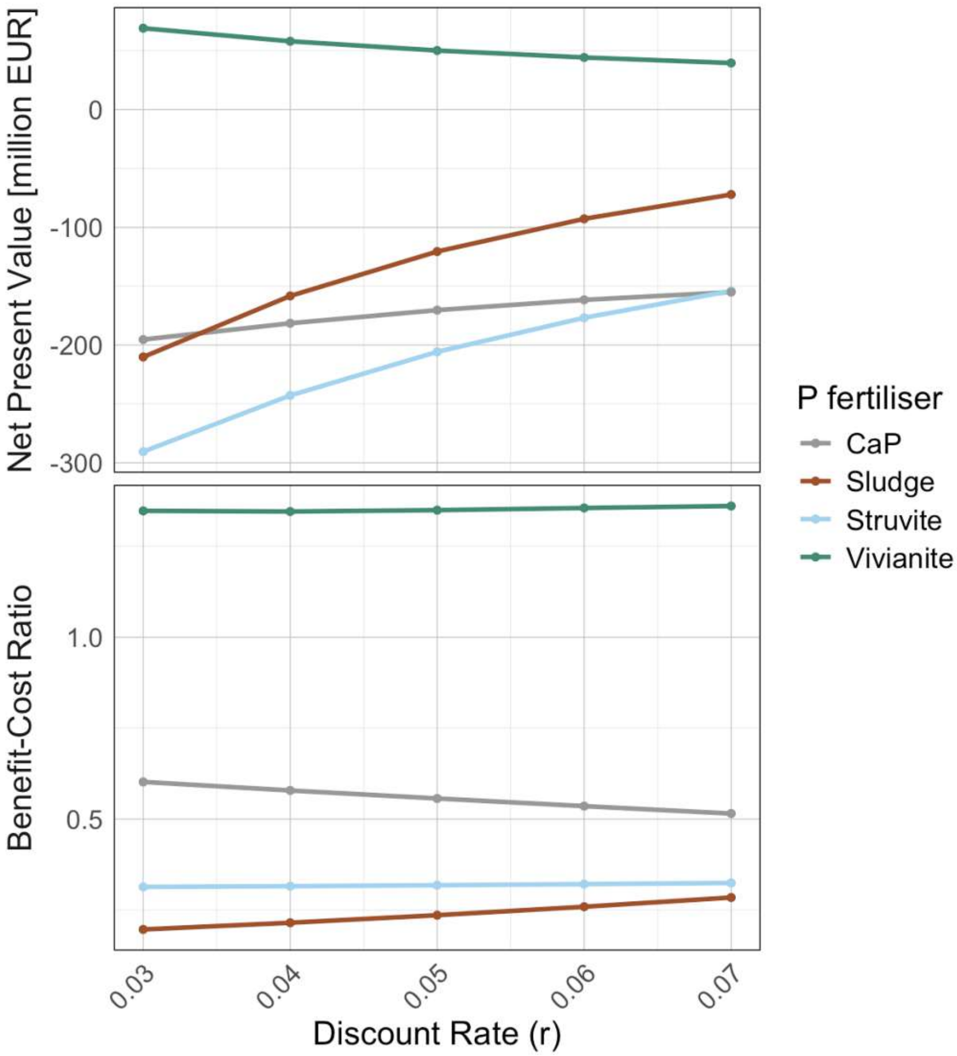

Figure 13. Sensitivity analysis of net present value (million EUR) and benefit-cost ratio for different discount rates for Catalonia in a time span of 50 years by treatment.

Our findings suggest that in some specific cases, P recycling technologies provide clear net benefits over direct sludge application, though their efficiency depends on technological recovery efficiency from sewage sludge, CAPEX, and long-term soil impact. Vivianite appears to be the most balanced option, offering both high recovery and low environmental externalities, although its agronomic use is limited by low solubility under non-flooded conditions. Struvite is preferable in small-scale or highly sensitive regions due to its safety profile, whereas CaP provides long-term sustainability advantages, but requires higher investments.

These insights support policy efforts aimed at accelerating the adoption of nutrient recovery technologies, especially in regions like Catalonia where heavy metal risks and plot prevalence are high. Still, marginal differences in NPV and B/C among P fertilisers (e.g., Valencian Community and Galicia) highlight the need to assess overlooked factors such as water pollution, soil degradation, and organic pollutants like PFAS and pharmaceuticals. Potential benefits such as additional nutrients and organic carbon soil input should also be considered to fully support decision-making. Incentivising such transitions through infrastructure subsidies, pollution-based taxes, or regulatory frameworks aligned with the Soil Health Law could ensure long-term agronomic viability and economic resilience in Spanish agriculture (*RD 1051/2022*, 2022).

4.5 Conclusion

This study provides an integrated assessment of the long-term environmental and economic implications of three advanced P recycling technologies (struvite, vivianite and CaP) compared to direct sewage sludge application in Spanish agricultural soils. By combining spatial soil data with a 50- and 100-year heavy metal accumulation model and cost–benefit analysis, we demonstrate that advanced P recycling can offer a potentially economically efficient pathway for mitigating heavy metal contamination while enhancing economic efficiency over time.

The results confirm that direct sludge application can lead to significant heavy metal accumulation, particularly of Zn, Cu, and Cd, posing risks to soil health, crop safety, and groundwater quality, especially in alkaline soils. In contrast, the recycled P products studied substantially reduce exceedance risks. Vivianite emerged as the most balanced option, combining relatively lower infrastructure costs compared to CaP, high P recovery efficiency compared to struvite, and low environmental externalities, though it has limited agronomic value.

CaP, in turn, requires higher capital expenditure but delivers superior long-term sustainability due to its negligible pollutant content and high recovery potential. Struvite, despite lower recovery efficiency, remains attractive for regions with small-scale agriculture or heightened environmental sensitivity. These environmental advantages are reinforced by the economic analysis. When accounting for indirect costs such as remediation and opportunity cost, both the economic net present value and benefit–cost ratios for recycled P options outperformed the baseline sludge scenario in some regions.

While our findings offer valuable insights, there are limitations, including assumptions of constant sludge application rates and the exclusion of metal leaching, crop uptake variability, and the potential agronomic benefits of organic matter and secondary nutrients. Additionally, the effects of emerging contaminants and broader environmental externalities remain insufficiently addressed, highlighting the need for further research. Despite limitations, this study presents an improved framework to integrate environmental externalities in the economic assessment to support sustainable agriculture while safeguarding long-term soil health at a regional scale. Future studies should refine these models by incorporating pollutant fate dynamics, crop-specific risks, and comparative life cycle assessments to inform more holistic nutrient management policies.

5. Summary and conclusions

5.1 The phosphorus challenge

Phosphorus is indispensable for food production, yet its current management is characterised by inefficiency, environmental damage and inequity. Phosphate rock reserves are geographically concentrated and subject to geopolitical risk, while large amounts of phosphorus are continually lost from the food system to water bodies and landfills, contributing to eutrophication and long-term ecological damage. At the same time, secondary and legacy phosphorus pools in manure, industrial wastes, wastewater and soils are estimated to exceed current annual phosphorus demand. Despite substantial technical progress in recovering phosphorus from these sources, global recycling remains marginal and most fertiliser demand is still met by primary mineral fertilisers.

Municipal wastewater emerges as a particularly promising phosphorus source in Europe. Wastewater treatment plants already concentrate phosphorus, and regulatory developments are increasingly restricting direct sewage sludge application, creating both a challenge and an opportunity. However, the deployment of wastewater-based fertilisers is constrained by strict quality requirements, heterogeneous agronomic conditions and economic uncertainties. This thesis responds to these tensions by investigating how improved utilisation of wastewater-derived phosphorus can be achieved when market-driven functions and requirements are explicitly aligned with the concrete properties of recovered phosphorus products, under real regulatory and agronomic boundary conditions.

Chapter 2 maps global secondary and legacy phosphorus resources and analyses the barrier landscape that currently prevents their large-scale use. Chapter 3 focuses on wastewater-derived fertilisers in Denmark, Germany and Spain, quantifying how much cereal phosphorus demand could be substituted under existing and emerging EU and national regulations and agronomic constraints. Chapter 4 then develops an integrated environmental–economic framework to assess long-term soil contamination risks and the economic performance of advanced recycling routes relative to continued sewage sludge application in Spanish agricultural soils. Together, these chapters provide a system-oriented assessment of how wastewater-based recycling can contribute to sustainable phosphorus utilisation in Europe.

5.2 Key findings

5.2.1 Global barrier landscape for phosphorus recycling

The first research paper (Chapter 2) demonstrates that secondary and legacy phosphorus resources are already sufficient to substantially support, or even exceed, current global phosphorus demand. Manure, mining and fertiliser industry wastes, wastewater, food-chain residues and accumulated stocks in agricultural soils and aquatic sediments constitute a resource base equivalent to more than current annual fertiliser demand. Yet these resources are largely unmanaged or underused, while phosphorus continues to be imported as mineral fertiliser and lost to sinks where it causes environmental harm. This reframes the phosphorus problem from scarcity to a problem of systemic inefficiency and poor governance of existing stocks and flows.

To explain this implementation gap, Chapter 2 develops a comprehensive typology of barriers to phosphorus recycling. Technical barriers encompass process reliability, scale-up challenges and product consistency, while knowledge barriers include limited understanding of product performance, legacy phosphorus mobilisation and long-term soil impacts. Logistical barriers relate to the spatial mismatch between waste generation and fertiliser demand, as well as low nutrient concentrations in many secondary streams. Economic and financial barriers arise from high capital costs, uncertain revenues and exposure to volatile fertiliser and energy prices. Societal barriers include risk perceptions, lack of familiarity among farmers and consumers, and contested acceptance of waste-derived fertilisers. Finally, regulatory barriers range from ambiguous waste status to fragmented or inconsistent quality requirements. These barriers act in combination along the value chain, creating a complex landscape in which recycled products often fail to meet the functional, economic or risk expectations of end-users and regulators.

Chapter 2 therefore argues that overcoming these obstacles requires integrated, systems-based and transdisciplinary approaches. Rather than focusing solely on individual technologies, governance interventions must simultaneously address information, incentives, infrastructure and institutions. The paper calls for closer collaboration between researchers, utilities, farmers, industry and policymakers, and highlights the need for decision-support tools that can make trade-offs transparent and align diverse stakeholder interests. This global framing motivates the subsequent focus on wastewater-based recycling in the European Union, where regulatory instruments, existing infrastructure and data availability make it possible to explore deployment under realistic boundary conditions.

5.2.2 Wastewater recycling potential under EU compliance and agronomic constraints

Building on this framing, Chapter 3 examines wastewater-derived phosphorus recycling in Denmark, Germany and Spain through the lens of legal compliance and agronomic suitability. The study first compiles relevant EU and national regulations, including the EU Fertilising Products Regulation, the Sewage Sludge Directive and national legislation in the three countries. These instruments define permissible pollutant concentrations, minimum phosphorus contents and, in some cases, technology-specific requirements. Out of 22 identified wastewater-based phosphorus recovery technologies, only a small subset of products fully complies with both EU-level and national fertiliser and soil protection rules across all three countries. Five technologies, representing calcium phosphate, vivianite and struvite products, are retained for further analysis. This finding illustrates how regulation not only safeguards soil and food quality but also strongly filters which recycling routes are realistically deployable.

The chapter combines information on wastewater treatment plant loads and configurations with the selected recovery routes to estimate regional potential phosphorus supply from each compliant product. Using data from the UWWTD Waterbase and national statistics, potential supply is expressed per capita and per agricultural area at the level of NUTS3 regions. The results show that potential wastewater-derived phosphorus supply varies widely between regions and technologies. Some regions, particularly those with large population centres and advanced treatment infrastructure, could generate substantial recycled phosphorus, while others have limited potential due to smaller treatment plants or lower connection rates.

To compare this potential supply with agricultural demand, Chapter 3 estimates regional phosphorus requirements for cereal production (wheat, barley and rye) under agronomically sound fertilisation strategies. This is done by combining crop area statistics with recommendations that distinguish between maintenance and build-up fertilisation, taking into account measured soil phosphorus status and pH from the LUCAS soil survey. The results show that available soil phosphorus is often above the levels required for optimal yields, indicating that a focus on maintenance or even drawdown strategies could reduce fertiliser inputs in many regions. Moreover, soil pH and phosphorus status constrain where particular recycled products can be applied without affecting soil or crop quality.

By comparing potential wastewater-derived phosphorus supply with regional cereal phosphorus demand, the study estimates the share of demand that could be covered by compliant recycled products under different deployment scenarios. While the specific contributions vary by country, technology and region, the results show that wastewater recycling can cover a substantial share of cereal phosphorus demand in some regions, while elsewhere its contribution is modest. In many cases, significant substitution would require redistribution of products from urban centres to cereal-producing regions, implying additional logistical and organisational efforts. Overall, the chapter demonstrates that wastewater-based recycling, when filtered through legal and agronomic constraints, represents a spatially heterogeneous but non-trivial opportunity to reduce dependence on mineral fertilisers and to limit sewage sludge application on agricultural soils.

5.2.3 Environmental and economic performance of advanced recycling vs sewage sludge

Chapter 4 complements the regional potential assessment by examining, in detail, the long-term soil and economic implications of substituting sewage sludge application with advanced wastewater-derived fertilisers in Spain. The chapter develops an integrated framework that couples a simple soil heavy metal accumulation model over 50- and 100-year horizons with a cost–benefit analysis that internalises selected environmental externalities. The framework compares continued sewage sludge application with the production and use of struvite, vivianite and calcium phosphate recovered from wastewater treatment plants that currently supply sludge to agriculture.

The heavy metal accumulation model assesses how repeated application of each phosphorus source affects soil concentrations of key metals (e.g. Zn, Cu, Cd) relative to Spanish legislative limits, distinguishing between acidic and alkaline soils and accounting for regional differences in climate and soil properties. The results indicate that continued sewage sludge application can lead to exceedances of heavy metal limits in a substantial proportion of plots, particularly in alkaline soils and in regions with high historic inputs. This poses long-term risks for soil health, crop quality and groundwater. By contrast, the application of struvite or calcium phosphate derived from advanced recovery processes results in far fewer exceedances, owing to markedly lower pollutant contents. Vivianite performs better than sludge but may still lead to localised exceedances for certain metals in specific regions, highlighting that not all recycled products share the same risk profile.

The cost–benefit analysis then evaluates the economic performance of each route over 50 and 100 years, including investment and operating costs, avoided sludge disposal costs, agricultural benefits, and the estimated costs of remediating soils where legislative limits are exceeded. While net present values are often negative across treatments when considering only direct costs and benefits, inclusion of environmental externalities substantially improves the relative performance of advanced recycling routes. In some regions, vivianite achieves positive net present values, reflecting its ability to recover a significant fraction of influent phosphorus at moderate cost while reducing long-term soil remediation needs. Struvite and calcium phosphate generally entail higher capital costs but offer clear environmental advantages and, in the case of calcium phosphate, very low pollutant contents and high recovery efficiencies. Overall, the analysis shows that when environmental externalities are explicitly accounted for, advanced recycling routes can be economically competitive with, and in some cases preferable to, continued sewage sludge application, even if direct financial returns remain modest.

At the same time, Chapter 4 acknowledges limitations such as the simplified representation of metal dynamics, the exclusion of potential benefits from organic matter in sludge, and the omission of emerging contaminants and full life-cycle impacts. Nevertheless, the framework illustrates how integrating long-term soil protection and environmental externalities into economic assessments can change the ranking of options and provide a more robust basis for decision-making on wastewater phosphorus management.

5.3 Converging functions and requirements with product properties

Taken together, the three research papers suggest that sustainable phosphorus utilisation depends on explicitly aligning market-driven functions and requirements with the concrete properties of recycled phosphorus products, under realistic regulatory, agronomic and economic constraints. Chapter 2 shows that large secondary and legacy phosphorus resources exist but are constrained by a web of interacting barriers spanning technology, economics, logistics, regulation and social acceptance. Chapter 3 demonstrates that, once legal compliance and agronomic suitability are enforced, only a subset of wastewater-derived products can be deployed at scale, and that their potential to substitute mineral fertilisers is unevenly distributed across regions. Chapter 4 then reveals that when long-term soil contamination risks and environmental externalities are integrated into economic assessments, advanced recycling routes can outperform conventional sludge application and offer robust options for soil and water protection.

Conceptually, this work moves beyond technology or purely regulatory perspectives by integrating global barrier analysis, compliance filtering, spatially explicit agronomic demand and environmental–economic assessment into a single narrative. It provides a template for assessing not just the technical feasibility but also the deployment readiness of wastewater-derived fertilisers. The results underscore that wastewater-based recycling cannot, and should not, be expected to supply all fertiliser needs. In turn, when their properties are aligned with functional requirements and embedded in supportive governance arrangements, advanced wastewater-derived products can form a key component of a diversified, circular phosphorus strategy that reduces environmental risks and improves resilience of fertiliser supply.

5.4 Implications for policy, practice and research

The findings have several implications for policy and governance. First, regulatory frameworks such as the EU Fertilising Products Regulation and national sewage sludge and soil protection laws play a decisive role in shaping which recycling routes are viable. Recognising advanced recycling technologies that demonstrably reduce long-term soil risks, and ensuring clear and consistent quality standards for recycled products, can provide a stable basis for investment and deployment. Second, national and regional phosphorus strategies need to account for the spatial mismatch between wastewater phosphorus supply and crop demand. This implies planning for logistics and redistribution, potentially supported by targeted economic instruments or infrastructure funding. Third, economic policies that internalise environmental externalities could help level the playing field between conventional sludge disposal, recycled products and mineral fertilisers.

For practice, the thesis highlights the need for coordinated action among utilities, technology providers and farmers. Wastewater treatment operators and technology companies must deliver products with reliable and well-characterised nutrient contents, low contaminant levels and forms that are compatible with existing fertiliser application equipment. Farmers need transparent information on agronomic performance in different soil and cropping systems, as well as assurance that recycled products comply with relevant regulations and do not compromise long-term soil quality. Demonstration projects, long-term field trials and knowledge exchange platforms can help build trust and familiarity with wastewater-derived fertilisers. Differentiated deployment strategies may be appropriate: for example, prioritising advanced recyclates in regions with high environmental sensitivity or where sludge-related soil risks are most acute.

Finally, the work points to important research needs. These include better quantification of legacy phosphorus mobilisation and its interaction with recycled inputs; more spatially detailed assessments that couple phosphorus flows, regulatory constraints and crop demand in other regions and for additional crops; and the integration of emerging contaminants, organic matter dynamics and full life-cycle impacts into comparative evaluations of recycling routes. More broadly, the thesis supports transdisciplinary research that actively involves stakeholders in defining relevant functions and requirements and in co-developing acceptable solutions. Advancing such approaches will be crucial to turning wastewater phosphorus from a liability into a cornerstone of resilient, circular and socially legitimate agricultural systems.

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List of abbreviations

Elements and molecules

Al	Aluminium
As	Arsenic
CaP	Calcium phosphate
Cd	Cadmium
Cr	Chromium
Cu	Copper
Fe	Iron
H_3PO_4	Phosphoric acid
Hg	Mercury
Ni	Nickel
P	Phosphorus
Pb	Lead
Zn	Zinc

Names

B/C	Benefit-cost ratio
CAPEX	Capital expenditure
CBA	Cost-benefit analysis
ESPP	European Sustainable Phosphorus Platform
EU	European Union
LUCAS	Land Use and Coverage Area Frame Survey
NPV	Net present value
NUTRIMAN	Nutrient Management and Nutrient Recovery Thematic Network
p.e.	Population equivalent
PR	Phosphate rock
PFAS	Per- and poly-fluoroalkyl substances
UWWTD	Urban Wastewater Treatment Directive
WWTP	Wastewater treatment plant

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Table A1. Heavy metal limits regulated in the sludge legislation to reuse for agriculture. [1]Values vary for different soil pH (Collivignarelli et al., 2019).

Pollutant [mg kg^{-1}]	Denmark	Germany	Spain[1]	EU
As	25	-	-	-
Cd	0.8	10	20-40	20-40
Cr (total)	100	900	1,500	-
Cu	1,000	800	1,000-1,750	1,000-1,750
Hg	0.8	8	16-25	16-25
Ni	30	200	300-400	300-400
Pb	120	900	750-1,200	750-1,200
Zn	4,000	4,000	2,500-4000	2,500-4,000

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	EU		**Denmark**		**Germany**		**Spain**	
	Name	**Remarks**	**Name**	**Remarks**	**Name**	**Remarks**	**Name**	**Remarks**
Collection	Urban Wastewater Treatment Directive (UWWTD) 1998 [98/15/EEC of 1998 amending 91/271/EEC 1991]	>2k PE has to be connected or treated	Waste Water Executive Order (WWEO) 2021 [BEK no 1393 of 21/06/2021]	Same as EU	Waste Water Ordinance (WWO) 2004	Same as EU	Royal Decree of Guidelines for Urban Waste Water Treatment (RDWWT) 1996 [RD 35/2023 of 23/02/2023]	Same as EU
Treatment	UWWTD 1998	WWTPs 10k-100k PE must comply with 2 mgP/L. >100k PE with 1 mgP/L (implemented in 2014)	WWEO 2021	WWTPs >5k PE must comply with <1.5 mgP/L. >100k PE with 1mgP/L	WWO 2004	WWTPs 20k-100k PE must comply with 2 mgP/L. >100k PE with 1 mgP/L[1]	RDWWT 1996	Same as EU
Outlet	UWWTD 1998	=	WWEO 2021	=	WWO 2004	=	RDWWT 1996	=
	Sewage Sludge Directive (SSD) (EC) 219/2009	Table A4	Sludge Order (SO) 2017	Table A4	Sewage Sludge Ordinance (SSO) 2017	Table A4	Royal Decree of Guidelines for Sludge Usage in Agriculture (RDSUA) 1990	Table A4
Usage	SSD 2009	=	SO 2017	=	SSO 2017	=	RDSUA 1990	=
	Market of Fertilising Products (MFP) 2019/1009	Table A2 and A3	Fertiliser and Soil Improvers Order (FSIO) 2009	Table A2 and A3	Fertiliser Ordinance (FO) 2012	Table A2 and A3	Royal Decree of Fertilising Products (RDFP) 2013	Table A2 and A3

*Table A3. Minimum phosphorus requirements according to fertiliser legislation. *Impurities refer to: organic matter, glass, stones, metal and plastics (Executive Order Act No 318 of 31 March 2007 on Fertiliser and Soil Improvers, Etc. 1), 2007; Real Decreto 506/2013, de 28 de Junio, Sobre Productos Fertilizantes, 2013; Verordnung Über Das Inverkehrbringen von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten Und Pflanzenhilfsmitteln (Düngemittelverordnung-DüMV), 2012).*

	Denmark	**Germany**	**Spain**	**EU**
Phosphoric Acid	-	8.7% P soluble in water	17.5% P soluble in water	2.2% P
Iron Phosphate	-	-	25% total Fe 8.7% P soluble in mineral acids	-
Precipitated Phosphates	-	4.4% P	-	7% P <3% organic carbon <0.3% impurities* <10% Al+Fe salts

Table A4. Heavy metal limits regulated in the fertiliser legislation and properties of phosphorus recycled technologies. 1 Values follow the class A/B/C labelling e.g., Cd 0.7mg/kg corresponds to class A. 2 For 3 mg/kg Cd with <2.2%P and 26.2 mg of Cd per kg of P with >2.2%P (Executive Order Act No 318 of 31 March 2007 on Fertiliser and Soil Improvers, Etc. 1), 2007; Real Decreto 506/2013, de 28 de Junio, Sobre Productos Fertilizantes, 2013; Verordnung Über Das Inverkehrbringen von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten Und Pflanzenhilfsmitteln (Düngemittelverordnung-DüMV), 2012).

Pollutant [mg kg^{-1}]	**Denmark**	**Germany**	**Spain[1]**	**EU**	**Ash2Phos® (*Ca-P*)**	**Crystal Green® (*struvite*)**	**PhorWater® (*struvite*)**	**ViviMag® (*vivianite*)**	**PAKU® (*ash*)**	**AshDec® (*ash*)**
As	40	40	-	40	1.4	<2	-	2	7	10.6
Cd	3/60[2]	1.5(2.5 non-food)	0.7/2/3	3/60[2]	<0.1	<0.4	<0.4	0.2	1.1	0.1
Cr (total)	-	300	70/250/300	-	1.7	<5	<3	16	170	90
Cr (VI)	2	2	Undetectable	2	-	-	-	-	-	-
Cu	600	900	70/300/400	600	5	-	<3	41	400	57
Hg	1	1	0.4/1.5/2.5	1	<0.1	-	-	-	<0.04	<0.1
Ni	100	80	25/90/100	100	2.5	<2	<3	9	97	0.9
Pb	120	150	45/150/200	120	3.6	<0.2	<2	12	15	12.3
Zn	1500	5000	200/500/1000	1500	-	<2	0.4-341	160	870	389

Table A5. Calculated potential P supply for all scenarios from general data of all regions in Denmark, Germany and Spain (Bundesanstalt für Landwirtschaft und Ernährung, 2019; Eurostat, 2022a, 2022b; Fernandez-Ugalde et al., 2022; Ministerio de Agricultura Pesca y Alimentación, 2022; StatBank Denmark, 2022a, 2022c; Urban Waste Water Treatment Directive, 2018). In Denmark, wheat, barley and rye cropland area (1,319,169 ha) corresponded at a country level to 92.9% (31.0%, 55.6% and 6.3%, respectively. In Germany, 48.9% covered by wheat, 27.1% by barley and 8.5% by rye, amounting to 84.5% (5,191,200 ha). In Spain, Cropland area of the three selected crops was 79.1% (4,767,230 ha), with 34.2% for wheat, 42.6% barley and 2.3% rye.

	Incineration	Total WWTPs	P-Removal WWTPs	P [tP]	Cereal cropland area [ha]	pH	Olsen P in soil [mgP/kgSoil]	POlsenS/ POlsenT Ratio	Population	Struvite	Vivianite Fe-P	Vivianite Ca-P	Ash Ca-P
DK-2018	12.8%	344	330	4,398	1,319,169	6.3	51.18	4.1	5,781,190	372.6	2,422.1	2,179.9	327.4
Capital		49	49	1,328	56,971	6.7	54.64	4.3	1,822,659	115.8	752.9	677.6	101.8
Zealand		75	72	531	284,044	6.9	39.15	3.1	835,024	44.5	289.0	260.1	39.1
Southern		99	90	801	350,536	6.3	51.49	4.1	1,220,763	63.5	412.9	371.6	55.8
Central		87	87	1,196	403,894	6.2	53.68	4.3	1,313,596	104.3	678.1	610.3	91.7
North		34	32	542	223,723	5.9	56.24	4.5	589,148	44.5	289.2	260.3	39.1
DE-2018	71.5%	3812	3247	67,330	5,191,200	6.7	58.19	4.6	82,792,351	1,556.5	10,117.2	9,105.4	23,405.2
Baden-Wurttemberg		610	566	8,451	367,500	6.8	49.78	4.0	11,023,425	223.6	1,453.7	1,308.3	3,363.0
Bayern		835	576	10,979	867,500	6.8	53.94	4.3	12,997,204	216.0	1,404.0	1,263.6	3,248.1
Berlin		2	2	2,199	-	0	0.00		3,613,495	62.7	407.7	366.9	943.1
Brandenburg		110	100	2,543	433,600	6.5	46.16	3.7	2,504,040	65.9	428.6	385.7	991.5
Bremen		4	4	546	-	-	-		681,032	15.6	101.2	91.1	234.2
Hamburg		1	1	1,400	-	-	-		1,830,584	39.9	259.5	233.6	600.4
Hesse		347	312	4,808	254,500	6.9	49.80	4.0	6,243,262	123.3	801.4	721.3	1,854.0
Mecklenburg-Vorpommern		94	74	1,286	526,700	6.7	47.18	3.7	1,611,119	28.9	187.7	168.9	434.2
Lower Saxony		426	406	6,998	705,800	6.2	85.57	6.8	7,962,775	190.2	1,236.4	1,112.8	2,860.4
North Rhine-Westphalia		492	479	16,575	434,600	6.7	63.94	5.1	17,912,134	460.2	2,991.6	2,692.4	6,920.8

Rhineland-Palatinate		311	273	2,823	193,000	7.1	47.59	3.8	4,073,679	70.7	459.4	413.5	1,062.8
Saarland		60	41	881	-	-	0.00		994,187	17.2	111.6	100.4	258.2
Saxony		155	102	2,640	335,900	6.8	46	3.6	4,081,308	49.5	322.1	289.9	745.1
Saxony-Anhalt		114	103	1,705	485,200	7	42.39	3.4	2,223,081	43.9	285.6	257.0	660.7
Schleswig-Holstein		134	120	2,228	265,400	6.6	82.49	6.5	2,889,821	56.9	369.9	332.9	855.7
Thuringia		117	88	1,268	329,300	7.1	39.06	3.1	2,151,205	27.2	176.8	159.1	409.0
ES-2018	4.3%	2335	813	20,838	4,767,230	7.7	24.94	2.0	46,658,447	892.5	5,800.9	5,220.8	241.4
Galicia		124	64	937	18,794	5.6	48.19	3.8	2,703,149	46.3	300.8	270.7	12.5
Principality of Asturias		32	13	258	85	6.9	48.45	3.8	1,027,624	10.0	65.2	58.7	2.7
Cantabria		26	10	102	839	6.2	63.60	5.0	581,294	3.7	24.3	21.9	1.0
Basque Community		47	13	572	36,065	8.1	17.76	1.4	2,170,868	15.1	98.4	88.6	4.1
Navarre		44	2	438	162,286	7.9	27.74	2.2	643,866	1.9	12.4	11.1	0.5
La Rioja		28	3	55	47,316	7.9	31.72	2.5	312,884	0.6	3.7	3.3	0.2
Aragon		108	20	650	725,663	8.0	23.31	1.9	1,313,135	11.5	74.9	67.4	3.1
Madrid		109	78	4,116	63,750	7.5	18.86	1.5	6,549,520	281.8	1,831.8	1,648.6	76.2
Castile-Leon		198	53	1,903	1,737,086	7.4	25.55	2.0	2,418,556	48.7	316.8	285.1	13.2
Castile-La Mancha		283	133	1,834	1,044,684	7.9	22.80	1.8	2,032,595	82.5	536.2	482.5	22.3
Extremadura		201	141	932	126,328	6.6	27.14	2.2	1,070,453	62.5	406.5	365.8	16.9
Catalonia		242	98	3,685	261,098	7.9	47.60	3.8	7,488,718	142.8	928.0	835.2	38.6
Valencian Community		255	101	1,863	19,129	7.9	46.54	3.7	4,946,233	70.6	459.0	413.1	19.1
Balearic Islands		81	12	542	26,164	-	-	-	1,166,923	7.7	50.0	45.0	2.1
Andalusia		418	38	1,385	467,571	7.9	17.11	1.4	8,410,095	12.0	78.3	70.5	3.3
Region of Murcia		48	31	1,529	29,757	8.1	23.18	1.8	1,475,569	94.5	614.0	552.6	25.6

Ceuta	1	-	-	-	-	-	-	85,209	-	-	-	-
Melilla	1	-	-	-	-	-	-	84,708	-	-	-	-
Canary Islands	89	3	38	615	-	-	-	2,177,048	0.1	0.8	0.7	0.0

Table A6. Calculated potential P demand, supply and share of demand covered by supply of most suitable recycled P technology for Denmark (vivianite Ca-P), Germany (ash Ca-P) and Spain (vivianite Fe-P), as well as the share of demand covered by supply if trade between regions with over 100% and neighbouring regions was enabled.

		P Supply [kg P/per cap.]	P Demand [kg P/per cap.]	Supply/Demand	Supply/Demand with trade	Trade remarks
DK-2018	Vivianite Ca-P in all	0.38	4.93	9%	-	
Capital		0.37	0.50	74%	74%	
Zealand		0.31	6.39	5%	5%	
Southern		0.30	5.24	6%	6%	
Central		0.46	5.50	8%	8%	
North		0.44	7.02	6%	6%	
DE-2018	Ash Ca-P in all	0.29	2.38	22%	-	
Baden-Wurttemberg		0.31	0.86	36%	36%	
Bayern		0.25	1.58	16%	16%	
Berlin		0.26	-	-	27%	
Brandenburg		0.40	2.89	14%	27%	With Berlin
Bremen		0.34	-	-	19%	
Hamburg		0.33	-	-	24%	
Hesse		0.30	0.94	31%	31%	
Mecklenburg-Vorpommern		0.27	6.58	4%	4%	
Lower Saxony		0.36	2.02	18%	19%	With Bremen
North Rhine-Westphalia		0.39	0.65	59%	59%	
Rhineland-Palatinate		0.26	1.19	22%	27%	With Saarland
Saarland		0.26	-	-	27%	
Saxony		0.18	1.85	10%	10%	
Saxony-Anhalt		0.30	4.36	7%	7%	
Schleswig-Holstein		0.30	2.14	14%	24%	With Hamburg
Thuringia		0.19	3.49	5%	5%	
ES-2018		0.05	1.18	10%	-	
Galicia	Vivianite Ca-P	0.10	0.08	148%	9%	
Principality of Asturias	Vivianite Ca-P	0.06	0.00009	74012%	9%	
Cantabria	Vivianite Ca-P	0.04	0.01	380%	9%	
Basque Community	Vivianite Fe-P	0.05	0.31	15%	15%	
Navarre	Vivianite Fe-P	0.02	3.92	0.5%	0.5%	
La Rioja	Vivianite Fe-P	0.01	2.79	0.4%	0.4%	
Aragon	Vivianite Fe-P	0.06	5.07	1%	1%	

Madrid	Vivianite Fe-P	0.25	0.11	259%	9%	
Castile-Leon	Vivianite Ca-P	0.12	9.94	1%	9%	With Galicia, Prin. Of Asturias, Cantabria and Madrid – Madrid with vivianite as Ca-P.
Castile-La Mancha	Vivianite Fe-P	0.26	4.95	5%	10%	With Valencian Community
Extremadura	Vivianite Ca-P	0.34	1.80	21%	21%	
Catalonia	Vivianite Fe-P	0.12	0.51	24%	26%	With Balearic Islands
Valencian Community	Vivianite Fe-P	0.09	0.03	349%	10%	
Balearic Islands	Vivianite Fe-P	0.04	-	-	26%	
Andalusia	Vivianite Fe-P	0.01	0.75	1%	11%	With Murcia
Region of Murcia	Vivianite Fe-P	0.42	0.08	513%	11%	
Ceuta	Vivianite Fe-P	-	-	-	-	
Melilla	Vivianite Fe-P	-	-	-	-	
Canary Islands	Vivianite Fe-P	0.0003	0.01	3%	3%	

Table A7. List of regions analysed for a combination of potential P supply and potential P demand (Table A6). Surplus regions are regions with potential P supply over 100%.

Receiving regions	**Supply/Demand**	**Supply/Demand with trade**	**Surplus regions combined**
Germany			
Brandenburg	14%	27%	With Berlin
Lower Saxony	18%	19%	With Bremen
Schleswig-Holstein	14%	24%	With Hamburg
Rhineland-Palatinate	22%	27%	With Saarland
Spain			
Castile-Leon	1%	9%	With Galicia, Prin. Of Asturias, Cantabria and Madrid – Madrid with vivianite as Ca-P.
Castile-La Mancha	5%	10%	With Valencian Community
Catalonia	24%	26%	With Balearic Islands
Andalusia	1%	11%	With Murcia

Table A8. Elemental composition of P-recycled products.

	Ash2Phos® (*Ca-P*)	Crystal Green® (*struvite*)	PhorWater® (*struvite*)	ViviMag® (*vivianite*)	PAKU® (*ash*)	AshDec® (*ash*)
Precipitated Phosphates [% total P]	16.5	12.2	11.8	11.5	4.7	8.5
Arsenic (As) [mg kg^{-1}]	1.4	<2	-	2	7	10.6
Cadmium (Cd) [mg kg^{-1}]	<0.1	<0.4	<0.4	0.2	1.1	0.1
Chromium (Cr) (total) [mg kg^{-1}]	1.7	<5	<3	16	170	90
Copper (Cu) [mg kg^{-1}]	5	-	<3	41	400	57
Mercury (Hg) [mg kg^{-1}]	<0.1	-	-	-	<0.04	<0.1
Nickel (Ni) [mg kg^{-1}]	2.5	<2	<3	9	97	0.9
Lead (Pb) [mg kg^{-1}]	3.6	<0.2	<2	12	15	12.3
Zinc (Zn) [mg kg^{-1}]	-	<2	0.4-341	160	870	389

Table A9. Price and yield data of crops for indirect cost estimation (Baruth et al., 2019; Ministry of Agriculture, 2018).

Crop	Freq	Price [EUR ha^{-1}]	Yield [t ha^{-1}]	Price [EUR]
Alfalfa	2	311	18	5,598
Almendro	2	0.0035	0.4	0.0014
Avena	1	255	2.6	663
Cebada	820	262	3.3	864.6
Centeno	5	247	2.8	691.6
Cereales de invierno para forraje	25	314	2.4	753.6
Hortalizas	1	11	39	429
Maíz	62	268	5.8	1,554.4
NULL	2			2558
Olivar	4		2.8	22,844
Otros cereales	12	291	1.8	523.8
Otros cultivos forrajeros	100	314	14	4396
Otros cultivos industriales	181	291	1.4	407.4
Otros cultivos leñosos	1	11	0.3	3.3
Trigo	1,223	391	3.7	1,446.7
Triticale	1	248	2.9	719.2
Viñedo	5	8	5.6	44.8

Table A10. Summary of treatment costs for each type of treatment for P supply (Egle et al., 2016; Ehsan, 2021; Kelessidis & Stasinakis, 2012; Uzkurt Kaljunen et al., 2022).

Treatment	Treatment cost [EUR t^{-1} P]	CAPEX [million EUR]	Remediation cost [EUR t^{-1} soil]	Transport cost [EUR t^{-1} P]
Sewage sludge (national average P concentration)	99	10 (250,000 PE)	100	9.1
Struvite	10	1.65 (500,000 PE)		
Vivianite	0.3	1 (100,000 PE)		
CaP	5	20 (1,750,000 PE)		

Supplementary figures

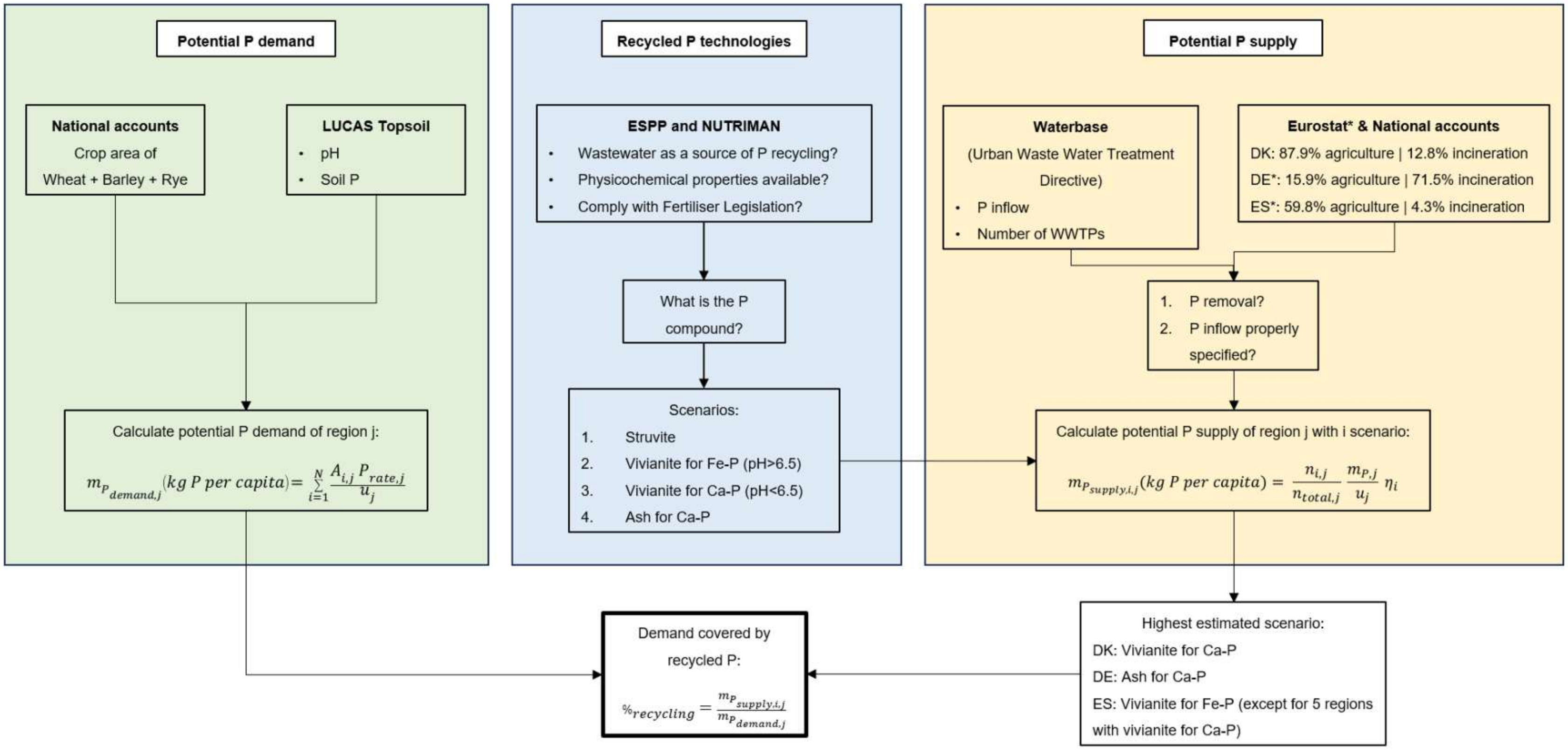

Figure A1. Flow diagram of methodology used to estimate potential P demand and supply from P recycling technologies available in the ESPP and NUTRIMAN catalogues. The estimated potential P demand and supply (the highest scenario), as well as the demand covered by the scenario with highest supply potential are specified per region in Table A6.

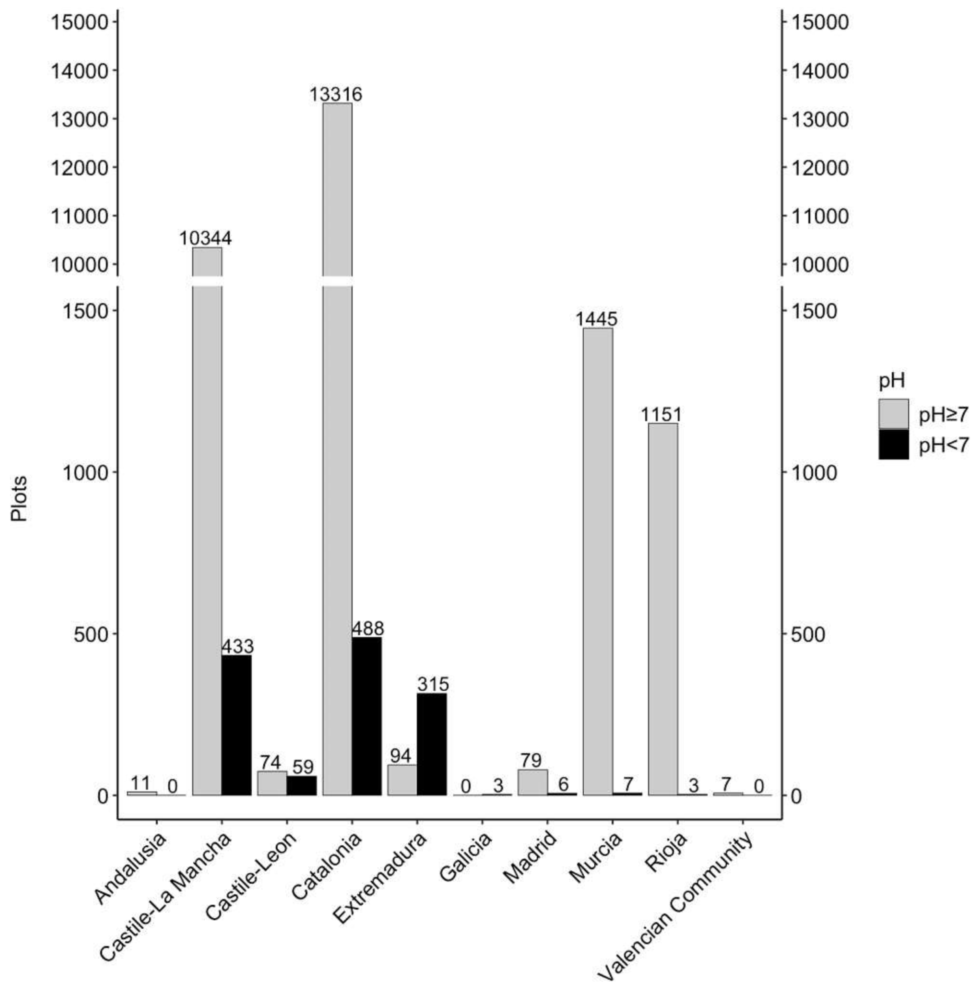

Figure A2. Distribution of agricultural plots across Spanish regions where sewage sludge is directly applied, grouped by soil pH.

Authorship

Chapter 2 of this thesis is based on the paper “Overcoming recycling barriers to transform global phosphorus management” by Henrique Rasera Raniro, Juan Serrano-Gomez, Harrie L. Mort, Teodor Kalpakchiev, Josephine Kooij, Yudong Zhao, Rodrigo M. Valença, Sinoxolo Magaya, Ana J. Guerrero-Esquivel, Leon Korving, Philipp Wilfert, Thomas Prot, Julia Martin-Ortega, Dorette S. Müller-Stöver, Mark van Loosdrecht, Dana Cordell, Jakob Santner, Henrikki Liimatainen, Ludwig Hermann, Matthew Scholz, Morten L. Christensen, Frederik van der Bom, Nelly S. Raymond, Søren Krogh Jensen, Fiona Smith & Kasper Reitzel.

This chapter has a shared first authorship between Henrique Rasera Raniro and Juan Serrano. The contribution of Juan Serrano on this work was:

- Project administration
- Conceptualization
- Visualization
- Data curation
- Writing of the original draft

Chapter 3 of this thesis is based on the paper “EU-compliant wastewater recycled phosphorus: how much national cereal demand can it meet?” by Juan Serrano-Gomez, Geneviève S. Metson, Tina-Simone Neset, Jakob Santner, Ludwig Hermann & Matthias Zessner.

The contribution of Juan Serrano on this work was:

- Conceptualization
- Methodology
- Formal analysis
- Investigation
- Visualization
- Writing of the original draft

Chapter 4 of this thesis is based on the paper “Integrated Framework to Assess Advanced Phosphorus Recycling as a Sustainable Alternative to Sewage Sludge in Agricultural Soils” by

Juan Serrano-Gomez, Henrique Rasera Raniro, Ludwig Hermann, Manuel Pulido-Velazquez & Matthias Zessner.

The contribution of Juan Serrano on this work was:

- Conceptualization
- Methodology
- Formal analysis
- Investigation
- Visualization
- Writing of the original draft

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