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Phosphorus recycling possibilities considering catchment and local agricultural system benefits: a review and regional Scottish case study

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Executive Summary

Research questions

1. What are the key factors affecting Phosphorus (P) material flows and recycling potential?
2. Is it feasible to develop a P flow analysis for Scottish catchments to explore local P capture and reuse?

Background

It is important to understand national to regional mass balances for key resources being tackled for sustainability, such as P, to consider local spatial aspects affecting resource recycling and reuse. Phosphorus sustainability couples material recycling with the dominant use as agricultural fertilisers, in turn with potential pollution of soils and waters by excess fertiliser usage, then as part of the P 'circular system', with P pollution from wastewaters resulting from human food consumption. This strengthens the need for planning units such as waterbodies and how they overlay with other boundaries, for example road distances. This project was developed to downscale aspects of Scotland's country-scale P mass balance by looking at the background context of, and a worked example of, catchment-regional recycled P sources, usage opportunities and constraints against the context of offsetting the raw imported resource of chemical phosphate fertiliser.

Research undertaken

- A literature review evaluated current knowledge on recycled-P fertilisers, covering production methods, agronomic benefits, environmental risks and policy framework underpinning their use.
- A case study catchment was selected based on (i) need to address river P pollution, (ii) presence of key nodes of P bearing resources that were current being, or could be, recycled, (iii) had an appropriate agricultural land bank for potential offsetting of chemical fertiliser.
- A framework was developed to analyse scenarios of P capture and reuse and applied for the catchment case study giving consideration of opportunities, potential conflicts and constraints such as costs (e.g. transport, capture, and processing costs).
- Recommendations are given for future projects on catchment-regional P budgets including transferable approaches, novel learning, gaps in research and ability to model processes.

Key Findings

1. The evaluation of the characteristics and properties of recycled P fertilisers showed that:

- There are many alternatives to phosphate rock (PR) sourced fertilisers. Recycled-P approaches all have strengths as well as weaknesses such as soil pollution.
- The greatest sustainability, agronomic and environmental benefits can be achieved by rather simple approaches with high P recovery potential, such as manures, composts and biosolids. Material use as fertiliser replacement balances suitability in P supply against risks associated with the potential of soil contamination, losses to water, crop/food contamination and GhG emissions.

2. Scenario modelling results showed that:

- Selection of a failing waterbody for P concentrations gave a central Scotland case study where an established 40,000 kgP/year of commercial anaerobic digestate and 24,000 kgP/year in farm-produced manures competed with the ability to incentivise P recycling (14,000 and 4000 kgP/year, from biosolids and final effluent) from wastewater. The mass and form of available P-bearing resources was challenging to assemble.
- Potential agricultural usage was considered according to crop fertiliser requirements and regulatory constraints (determined by material compositions and processing level). Fertiliser P dosage levels were attained before soil metal pollution thresholds were exceeded.
- Fertiliser scenarios comprised: (i) chemical fertiliser (reference), (ii) accredited digestate use, (iii) wastewater sludge use, and (iv) struvite use, for (a) the waterbody only area (592 fields) and (b) by progressively using fields in successive 10 km road transport bands (to a subsidised transport distance of 60 km; 110K fields). Deficits and excesses of manures had a strong bearing on recycling opportunities in this mixed farming area. There was minimal potential for recycling in the waterbody area and the materials were usable within 20 km road distance (12K fields). A hypothetical processing scenario of struvite production from wastewater gave a 1600 kgP/year contribution, reducing effluent P concentrations and being cost effective as a fertiliser.
- A simple diffuse pollution risk framework was applied based on relative risks of riparian versus non-riparian fields and relative erosion risk of landscape-crop combinations. Few fields (~20%) were adjacent to watercourses and in high-risk classes for erosion (~10%).

Recommendations and policy implementation

- The large workload of identifying the available P resources showed a lack of coordination of inventories of waste-materials available to manage resources such as P sustainably.
- Despite a motivation to explore P recycling against water pollution benefits (wastewater P recovery) an excess of P bearing materials for the waterbody led to exploring wider farmed areas. Hence, effective P planning requires joining up planning scales and boundaries, not just those of catchments (e.g. road networks, fields-farms, local authorities and waterbodies).
- Urbanised areas have abundant P beyond local land bank requirements, with established industries (e.g. anaerobic digestion) subsidising P distribution to farmers in competition with incentivising wastewater P recovery. Hence, cross-sectoral coordination is required.
- Effective planning must ensure (i) recycled P-bearing materials offset chemical fertiliser at agronomic best practice constraints and remove excessive disposal onto 'sacrificial' and polluting localised areas and (ii) that farm manures are used effectively in priority.
- Advanced processing of P-bearing materials is scarcely practised in Scotland but some of these (e.g. struvite recovery from wastewater) have potential to maximise the ability to use recovered P over increased transport distances and over wider crop types (struvite is P dense for transport and sufficiently pure to remove crop restraints).
- Research is needed to develop models to understand the P diffuse pollution implications of changing chemical P fertiliser to partial or complete replacement with alternative materials. Simplistic assumptions (e.g. transferring manure-based mineralisation rates) are not robust to the growing diversity of potential soil amendments. New approaches can utilise existing approaches for model application (e.g. national soil-landscape risk maps, diffuse pollution approaches developed for waste licencing), but need new data on P solubility and leaching.

1. Rationale for project

The project builds on a national phosphorus resource budget developed in the CREW project "Water Resource Balancing: Is a Closed Loop System Possible that Enhances Sustainable Rural Supplies" (Hough et al, 2016). It was recognised that there is considerable spatial coupling of the production and utilisation of many of the resources capable of supplying recycled-Phosphorus (P) in different forms. This arises from distances between centralised nodes of resource handling (often near population centres) and dominant use areas for chemical fertiliser replacement coupled with declining sustainability of transport to land over longer distances. The current project aimed to map potential pathways for P flows for example Scottish catchments. The original objectives of the current work were:

- Review factors affecting P material flows and recycling potential.
- Produce a P flow analysis for case study Scottish catchments of contrasting land use.
- Analyse several scenarios (e.g. P recapture using different strategies, reuse in various locations) for the two catchments and identify areas that deliver best practise in P management and offer opportunities for generating value. Explicitly consider the costs associated with the proposed opportunities (e.g. transport, capture, and processing costs).
- Provide recommendations for future projects on regional P budgets.

Consultation with the project steering group revealed key national policy motivations for such work. It was envisaged originally that this project would look at the spatial aspects of linking a land bank (with opportunities and constraints in conventional and replacement fertiliser usage) with local sources of P that may potentially be reused. It was recognised that P security and sustainability issues of reuse and recycling are topical issues in Scotland as they are in other EU countries. However, the case was made that current economics associated with replacement of chemical P fertiliser with alternative materials does not itself provide sufficiently strong argument for recycling and reuse. Instead, the current policy driver is to remove P from waste streams that pollute the water environment and are causing downgrades of water quality under Directive 2000/60/EC, aka the Water Framework Directive (WFD). Such pollution is recognised to include both direct discharges to water (e.g. point source effluents) or excess local application of by-products (e.g. anaerobic digestate) onto already P-enriched soils (promoting diffuse pollution). This resulted in the project having a catchment basis where site selection started with screening catchments failing for P status under national WFD monitoring and having several nodes of P processing. Accordingly, it was recognised in this project that P reuse in agriculture (as a dominant reuse potential) is currently made more viable where the combined system of intervening in P-rich wastes and utilisation on land contributes to the goals of improving compliance in the WFD status of local freshwaters.

2. Literature review

2.1. The context of P resource security

2.1.1. Rock phosphate resources and food production

Phosphorus (P) is an essential element for all life forms and is often the limiting nutrient in agricultural soils. In the past, P in crop residues, guano, bone meal, animal manure and human faeces was returned to the soil to aid in crop production (Ashley et al. 2011). However, without the discovery of the commercial manufacturing process for chemical P fertilisers from phosphate- rich sedimentary rocks, aka phosphate rock (PR), agricultural soils would grow steadily less productive under the pressure of agricultural intensification and population growth (Dawson and Hilton 2011; Daneshgar et al., 2018).

P losses occur at every stage of the food system, with one-fifth of the phosphorus mined for food production finding its way into the food consumed by the global population each year (Cordell et al., 2009). In some regions of the world, including Scotland, past use of P fertiliser and inefficient soil management led to soil P accumulation to levels that exceed crop requirements or in chemical forms that are unavailable to crops. This resulted in widespread P losses from soil to water systems leading to endemic aquatic eutrophication, reduced biodiversity and poor drinking water quality (Frossard et al., 2009; Sharpley et al., 2013; Cordell and White 2014; Toth et al., 2014; MacDonald et al., 2016;); see also Section 3.1. Once P is transported to water systems, it is not recoverable in the foreseeable future (Smit et al., 2009). The combination of modern agriculture's strong dependency but inefficient P use and the finiteness of PR deposits recently attracted increased scientific and public attention into the potential and risks of P scarcity.

It is widely recognised that in a world of 9.7 billion people by 2050 (United Nations 2019), securing enough of essential, non-substitutable P for plant growth is key to meeting livestock and human dietary requirements. After the 1950's, PR fertiliser became the primary P source to agricultural land, with 80- 90% of global PR reserves currently being used as fertilizers for crops and fodder

(Cordell et al., 2009; Daneshgar et al., 2018). There are two major approaches to estimating PR reserves: the ***Peak P mode,*** which predicts that PR production would peak in and the 2030-2035 followed by depletion by 2100 (Cordell et al., 2009; 2011); and the ***Lifetime of available reserves model,*** which assumes that the rate of P consumption will regulate the rate of PR reserve depletion, and therefore there is no imminent risk of PR shortages (van Kauwenbergh 2010) (see more in Appendix I.1).

Estimating PR reserves and resources is extremely complex and depends heavily on the interplay between demand and supply (Giraud 2012; Scholtz and Wellmer 2019) and on the cost-efficiency of innovation in the mining and extraction processes (Clift and Shaw 2012; Daneshgar et al., 2018; Geissler et al., 2018). Further, there are issues with the uneven distribution and quality of PR reserves, which raise issues of food security for PR importing countries in the face of geopolitical risks and heavy metal impurities (Mortvedt and Sikora, 1992; Kpomblekou and Tabatabai, 1994; De Ridder et al., 2012; Steiner et al. 2015). Appendix I.1 discusses in further detail the two approaches to estimating PR reserves, and the implications related to their uneven distribution and quality.

2.1.2. Current state of concepts for resource efficient P usage in agri-food and (waste) water sectors

The circular economy (CE) concept is gaining popularity among political and economic decision-makers over the currently predominant linear economy model of take-make-use-dispose. Boulding (1966) first introduced the concept of closed systems envisaging a future economy that would operate by supplementing the limited stock of inputs by recycling waste outputs. Examples of how P management fits into some CE concepts are discussed below.

Industrial ecology considers that the natural ecosystem and man-made industrial system operate in a similar way and are characterised by flows of materials, cash, energy and information (Garner & Keoleian, 1995; Erkman, 1997; Ehrenfeld, 2007). For P this includes mapping P flows globally (e.g. Clift and Shaw 2012; Bouwman et al. 2013; Scholz and Wellmer 2015) and for specific areas, such as for the US (Suh and Yee 2011), China (Chen et al., 2008), Australia (Cordell and White 2011), UK (Bateman et al. 2011; Cooper 2015) and Scotland (Hough et al, 2016; this example for a Scottish-based P flows analysis is given in Box 1). P flow mapping diagrams show dominant contributions of sources of phosphorus to overall flows starting at raw resource levels up to receiving environmental compartments (waters, soils and sediments), agriculture, food production and wastes. P flow mapping has shown that overall, the use efficiency of mineral P (i.e. the ratio between the amount of P in digested human food and the amount of P in PR mined for fertiliser use and feed additives) is in the range of 5 to 10%. This low P use efficiency is due to P losses along all stages of the supply-demand chain from PR mines to manure management and wastewater disposal. Studies using P flow mapping conclude that: (i) the majority of the P entering the global food production system is lost into water rather than remaining concentrated as recoverable waste; and (ii) the greatest opportunities for improving P use efficiency and reducing environmental impacts lie in reducing meat and dairy, enhancing P recovery from waste and limiting soil P accumulation.

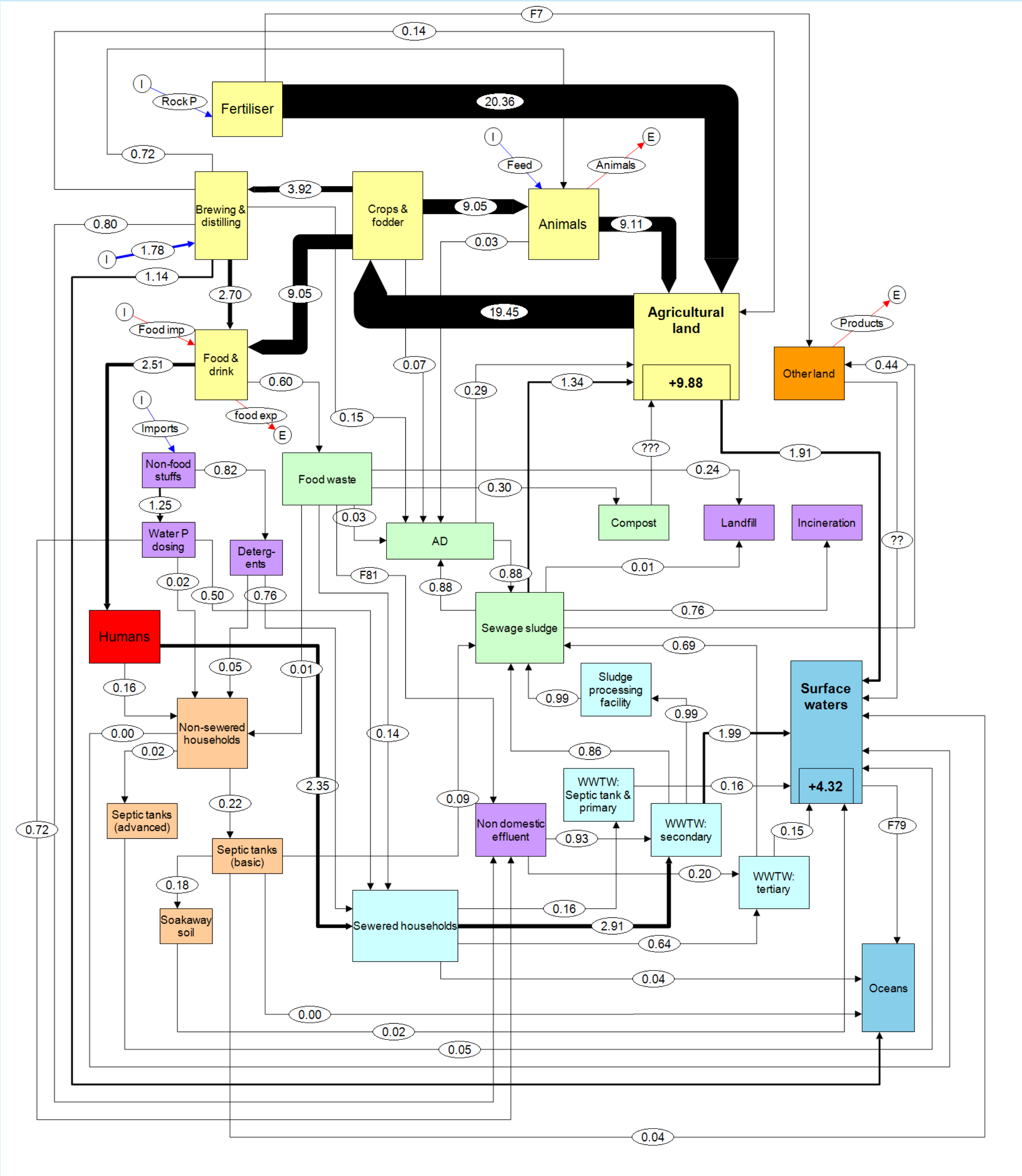
Industrial symbiosis applies industrial ecology principles at the company level to foster synergistic collaboration within and between companies on exchanging resources and by-products (Chertow, 2000) to promote eco-innovation (Lombardi & Laybourn; 2012). As an example, industrial symbiosis approaches may further P recovery through integrating processes around handling of anaerobic digestate (AD) and municipal solid waste (MSW) to achieve products out of what was previously considered as waste. This may be achieved via integrations of AD and liquid digestate use, AD and digestate pyrolysis, or AD and digestate incineration (Peng and Pivato 2019; Monlau et al., 2016).

"Cradle-to-cradle" design and "Zero Waste" strategies seek to maintain and enhance the value, quality, and productivity of material resources to bring net positive environmental effects (Braungart et al., 2007; Ankrah et al., 2015). This contrasts with traditional sustainability approaches (e.g. industrial ecology) that focus on reducing or eliminating the negative environmental impact of human activity. The "cradle-to-cradle" philosophy has given rise to "Zero Waste" strategies around the world such as separating waste types at collection, reuse, recycling, composting, repairing materials and involving the civil society in resources and waste management (de Jesus et al., 2018); see also the "Circular Economy Package" in the EU (European Parliament 2018)

Life cycle assessment (LCA) for P fertilisers can assess the environmental impacts associated with all stages of P production from PR mining or P recovery from recycled material through fertiliser manufacturing, distribution and use (Bradford-Hartke et al., 2015; Cieslik and Konieczka 2017; Kalmykova et al., 2015; Linderholm et al., 2012; Möller et al., 2018; Nakakubo et al., 2012; Sena and Hicks 2018; ten Hoeve et al., 2018).

The LCA approach uses well-established methods such as ISO 14040 and ISO 14044 (Finkbeiner et al., 2006; Baumann and Tillman 2004) to conduct holistic environmental evaluations comparing the environmental impact of different options. To provide a common basis

Box 1. A country-level P flow assessment for Scotland developed in an earlier CREW project (Hough et al. 2016). The analysis was undertaken using 2015 data derived from a literature review, primary data, expert input and modelled estimates. Arrow widths (and values) represent flows (in ktonnes P year^{-1}) between the considered system components (boxes, coloured: yellow, agronomic system; purple, industrial components; green, recycling opportunities; mid blue, surface waters; light blue, centralized wastewater systems; tan, decentralized wastewater systems). Increases in stocks are shown for agricultural land and surface waters. Letters 'I' and 'E' denote imports and exports, respectively, which have not been quantified. The results are further discussed in section 2.3.1 of the present study.



for consistent, robust and quality-assured life cycle data and studies, the International Reference Life Cycle Data System (ILCD) (European Commission 2010 cited in ten Hoeve et al., 2018) has recommended the following broad environmental impact categories: depletion of reserve-based abiotic resources, freshwater eutrophication, climate change, depletion of fossil abiotic resources, ecotoxicity, human toxicity, carcinogenic, non-carcinogenic, marine eutrophication, terrestrial eutrophication, terrestrial acidification, stratospheric ozone depletion, particulate matter formation, ionizing radiation and photochemical oxidant formation.

An example (taken from Möller et al., 2018) for the processing of different P waste streams is given in Table 1. This example uses five major impact categories for assessing the environmental impacts of P fertilisers:

1. Finite abiotic resources depletion potential (ADP) in kg antimony equivalents (Sb-eq); this refers to impacts on the availability of PR reserves.
2. Primary energy demand (FED) in mega joule (MJ); this refers to impacts on depletion of fossil fuel resources.
3. Global warming potential (GWP) calculated as carbon dioxide equivalents (CO_2-eq) with a time horizon of 100 years; this refers on greenhouse gas (GhG) emissions during production, transport, and application of P fertilisers.
4. Acidification potential (AP) calculated as sulphur dioxide equivalent (SO_2-eq); this refers to emissions of compounds that are precursors to acid rain, such as sulphur dioxide (SO_2), nitrogen oxides (NOx), and other various substances.
5. Eutrophication potential (EP) calculated as phosphate equivalents (P_2O_5-eq); which usually refers to global P losses to water, or to freshwater systems[1]

1 This is based on the simplifying assumption that P is limiting in freshwater ecosystems and N is limiting in marine ecosystems (see review by Morelli et al., 2018).

The LCA studies (see below) indicate strong differences among waste streams and treatment approaches. Despite differences in the choice of impact categories and indicators, and system boundaries between studies, a pattern has emerged, as follows:

- Simple approaches (e.g. direct application of biosolids) provide more favourable environmental impacts in all five major LCA impact categories (see Table 1) than more sophisticated ones (e.g. struvite production). The comparatively better performance of directly applied biosolids is related to: (i) the higher energy and chemical resources consumption of P recovery processes such as chemical precipitation or incineration; and (ii) reduction of fertiliser effectiveness and organic matter value following chemical or thermal processing of biosolids, thereby reducing the potential for carbon sequestration and increasing losses to the water environment.
- Anaerobic digestion (AD) results in lower GHG emissions than composting and therefore it has a more favourable environmental performance. An additional energy gain for AD is its contribution to offsetting fossil fuels due to produced biogas from AD and the higher nutrient losses from composting. However, it is recognised that that the potential soil carbon sequestration per unit P is higher for composts than for digestates.

Table 1. Life cycle Assessment of key recycled-P production methods in relation to P recovery rates with an evaluation of overall recycling potential using PR and TSP as controls (modified from Möller et al., 2018).

	Life cycle assessment indicators						
	ADP	**FED**	**GWP**	**AP**	**EP**	**Recycled-P recovery**	**Overall recycling potential**
Approaches of P recovery per kg P	Sb-eq /kg P	10 MJ/kg P	CO2-eq/ kg P	0.01 SO2-eq/kg P	0.01 P205-eq/kg P	rate	
PR	⇧	⇔	⇔	⇔	⇔	Control	Control
TSP	⇧	⇧	⇧	⇧	⇧	Control	Control
Organic household (biomass) waste compost	⇔	⇩	⇧	⇧	⇧	Very High	Intermediate
Organic household waste digestate	⇩	⇩	⇩	⇩	⇩	Very high	Intermediate
Biomass ash-untreated	⇩	⇩	⇩	⇩	⇩	Very high	Low
Biomass ash-solubilised	⇧	⇧	⇧	⇧	⇧	Very high	Low
Slaughterhouse waste (dried)	⇩	⇩	⇩	⇩	⇔	Very high	Intermediate
Slaughterhouse waste (digested)	⇩	⇩	⇩	⇩	⇩	Very high	Intermediate
Biosolids (dewatered sewage sludge)	⇩	⇩	⇩	⇩	⇩	Very high	High
Untreated sewage sludge ashes	⇩	⇩	⇧	⇔	⇔	Very high	High
Solubilised sewage sludge ashes	⇧	⇧	⇧	⇧	⇧	Very high	High
Struvite	⇧	⇧	⇧	⇧	⇧	Low	Intermediate

ADP: Abiotic resources depletion potential; FED: Primary energy demand; GWP: Global warming potential; AP: Acidification potential; and EP: Eutrophication potential.

⇔ no contribution ⇩ reduction ⇧ increase

- Solubilisation of ashes results in stronger environmental impacts than PR solubilisation to obtain water-soluble triple super phosphate (TSP). P concentration in PR is much higher; therefore, the consumption of finite abiotic resources and fossil fuels and the emissions to air and water per unit P are higher.
- Composts, chemically solubilised ashes from sewage sludge and struvite precipitation represent P recovery processes with the highest GHG emissions.
- Composts, chemically solubilised ashes, struvite and TSP have the greatest reactive-P losses to water.

Agronomic benefits of different P waste streams and the environmental impacts arising from their chemical and microbiological content are reviewed in the following sections in the context of existing LCA assessments (e.g. Table 1) and waste management policies and regulations. The findings are summarised in Table 6 (Section 2.4) to help better understand benefits, practicalities and risks of P recycling approaches.

2.1.3. Current state of actions in Scotland for resource efficient P usage in agri-food and (waste) water sectors

Major drivers for domestic policy on CE come from the EU (Korhonen et al., 2018). In 2018, the European Commission adopted a CE work package indicating that: "the transition to a more circular economy, where the value of products, materials, and resources is maintained in the economy for as long as possible, and the generation of waste minimized, is an essential contribution to the EU's efforts to develop a sustainable, low carbon, resource efficient, and competitive economy" (European Parliament 2018). The EU recognises CE benefits for the environment, security of supply of raw materials (including PR), competitiveness, innovation, growth and jobs. However, it also recognises challenges for financing, waste governance, consumer behaviour and business models of waste management. To overcome these challenges, the Commission adopted the Circular Economy Package and the Circular Economy Action Plan (Appendix I.2) to give a clear signal to economic operators that the EU is using all the tools available to transform its economy, opening the way to new business opportunities and boosting competitiveness.

In Scotland, legislative provisions and requirements regarding P have developed in parallel and continue to adapt to the EU's Circular Economy Package and Action Plan. Specific initiatives enabling P recycling and recovery from P-containing waste in Scotland include the "Zero Waste Plan" and the "Making Things Last" strategy. These are discussed in Appendix I.3.

2.2. Environmental issues of phosphorus and management

2.2.1. Phosphorus (P) as an environmental pollutant

P is both a critical resource and a potential pollutant of global concern and impact. Phosphorus is added to and cycles through various environmental compartments through natural and anthropogenic P cycles, involving numerous P transfer between phosphate-bearing rock deposits, soil, organisms, water, sediments and humans (Ruttenberg 2003; Filippelli 2008; MacDonald et al., 2016). Suggestion. The global natural and anthropogenic P cycles are described in Appendix II.1.

Direct transfer of untreated or partially treated human waste to waterbodies via sewerage systems, and P leaching and P losses in surface runoff from poorly managed P-rich agricultural soils are the two most significant processes rendering P an environmental pollutant and a threat to the water environment (Withers and Bowes 2018). Sewage P pollution pressures are more significant in urban catchments; agricultural P pressures are more significant in rural catchments. Once P concentration in surface waters exceeds a certain threshold relative to certain sensitivity factors for a specific ecosystem, it can trigger a range of cascading outcomes known as eutrophication. The following outcomes are signs of eutrophication: excessive algal blooms and plant growth, microbial community shifts towards cyanobacteria, high water turbidity, loss of macrophytes as they become shaded-out by algae, low-oxygen conditions and wildlife loss, e.g. fish kills. Eutrophication increases water treatment costs to remove algal toxins and decomposition products of algal biomass and reduces the value of recreational and residential areas due to "murky waters", which are perceived as a health risk (Dodds 2007; Smith and Schindler 2009).

2.2.2. Phosphorus (P) accumulation in soils and resulting losses to waters

In agricultural areas, P losses to the water environment increase with P soil saturation (Nair 2014; Simmonds et al., 2015). Plants take up large amounts of P from the soil solution as phosphate (P_2O_5) anions. P_2O_5 concentration in the soil solution is normally very small (typically 10^{-5} M; Roberts and Johnston 2015). This is due to strong P affinity for complexation by soil geochemical surfaces and soil biogeochemical transformations. Therefore, P may be unavailable to plants during the growth period, but too much P in the soil is a sign of inefficient P use and increases the risk of losses to the water environment. See Appendix II.2 for a summary of key soil biogeochemical transformations determining P pools and P plant availability

Intensive application of PR-derived fertiliser to meet crop requirements has led to the accumulation of P in agriculturally managed soils (legacy soil P). In these soils the ratio of inorganic P (partly available/partly immobilised P[2]) to organic (immobilised) P in the soil has shifted towards inorganic P (Negassa and Leinweber 2009; Stutter et al., 2015). This increases the risk of dissolved inorganic P losses to groundwater through leaching as well as of inorganic dissolved and soil-bound P losses in farmland runoff. Organic matter (OM) - rich soils, and peat soils (OM concentration >20%) have much lower capacity to retain P (Wall & Plunkett, 2016). Surplus inorganic P, in excess of crop requirements, is more likely to be lost, especially during periods of high rainfall and when water begins to drain from high OM soils. Algae in rivers, lakes and estuaries are extremely efficient at assimilating soluble inorganic P, hydrolysing organic forms of P and utilising soil particle-bound P when other forms of P are limiting (Heisler et al., 2008; Camarero and Catalan 2012).

Achieving the right balance between providing the right type and amount of P fertiliser to meet crop P requirements and preventing P losses in runoff or leaching to protect the environment from eutrophication and to reduce PR imports, is a key CE objective. Achieving this objective is vital in areas where there are additional P pressures on waterbodies from a wide range of sources including the food industry, wastewater treatment works (WwTW), septic tanks and livestock (Withers et al., 2015; Withers and Bowes 2018). Therefore, it is important to consider the composition and availability of P and efficiency of PR-derived and recycled-P fertilisers to identify which type of P recovery process and source materials deliver a P fertiliser that is fit for purpose in balancing crop requirements and environmental losses.

The current state of the agronomic - environmental P system leaves considerable potential for interventions to improve efficiency. These are demonstrated using Figure 1 against the context and mass balances of a model global arable farming systems (derived from mixed literature data sources – see Fig. legend). The average P flows in this simplified system show that global systems utilise manures well with an equivalent amount of manure to chemical fertiliser being applied to arable land. However, the fate of the sum of 29 TgP/year of fertiliser and 20 TgP/year of rock-derived P weathered in-situ, only produces 10 TgP/year of crop production (7 TgP/year useable if yield losses are accounted for) and 12 TgP/year of animal production,

2 In organic matter (OM)-poor soils, inorganic P bound to Fe + Al (hydr)oxides can make up a large proportion of the immobilised P.

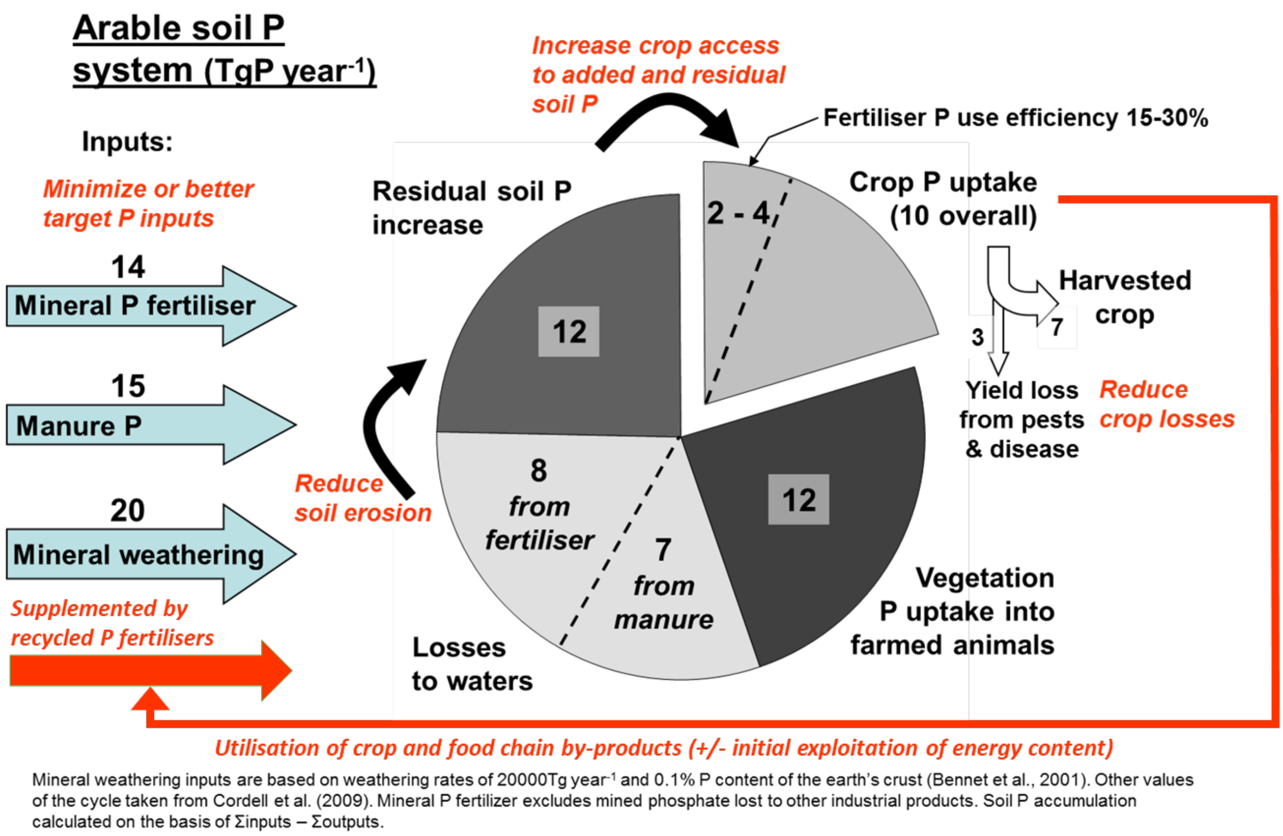


Figure 1. Model arable farming system using average global data to demonstrate system inefficiencies with respect to phosphorus and intervention opportunities. Mineral weathering inputs are based on weathering rates of 20000Tg year-1 and 0.1% P content of the earth's crust (Bennett et al., 2001). Other values of the cycle taken from Cordell et al. (2009). Mineral P fertiliser excludes mined phosphate lost to other industrial products. Soil P accumulation calculated on the basis of Σinputs – Σoutputs.

whereas accumulation in soil beyond use by current crops and losses to water constitutes 12 and 15 TgP/year, respectively. Examples of positive interventions are shown in red in the figure. These include agronomic goals of better spatial and temporal targeting of fertiliser inputs, reductions of P losses from fields, improving P acquisition by crops and minimising harvest losses by better pest and disease control. The 14 TgP/year that comprises chemical (largely PR-derived) fertiliser may be offset by use of recycled-P materials as fertiliser alternatives and this can include 'circularity' by use of by-products of the crop and food chain (perhaps following energy exploitation e.g. in the form of digestate following biogas production). However, alternative materials must satisfy conditions of not increasing diffuse pollution P losses, making P less available to crops or adding to harvest wastage (for example limiting markets due to social acceptance or microbial or trace chemical contamination) (for further discussion see Stutter et al., 2012).

2.2.3. Phosphorus discharges from wastewaters and other point pollution sources

Since the mid-20th century rapid food production and urbanisation led to significant eutrophication of freshwaters by P. During the 1970's to 1990's a large portion of the P loading was addressed by implementation of legislation and actions for effluents, urban runoff and industrial pollution. Since then, diffuse pollution, including polluted runoff from fields, farmyards and rural unsewered wastewaters, has been a necessary focus; but one that is much harder to identify and manage than point source pollution (Le Moal et al., 2019). Although the major P loadings to rivers have been greatly reduced and water quality has become much better in the last couple of decades there remains a threat of aquatic ecosystem damage associated with P and eutrophication. In many cases improvements in chemical P concentrations have not been followed by an expected rate of improvement in ecology and this has many complex multiple stressor aspects behind it. However, in terms of P source loadings wastewaters remain a significant threat to river ecological status acting alongside these other stressors such as flow regulation, elevated temperatures and emerging contaminants. Also, a considerable amount of P that may be captured and potentially recycled remains lost to waters. The P budget study shown in Figure 2 concluded, at the time in 2011, that discharge to waters accounted for 43% of 55 ktonnes P/year that was the sewage P budget at a UK level. In a separate analysis White and Hammond (2009) suggested that a modelled 60ktonnesP/year total P load to GB waters was distributed 73% domestic sewage, 20% agriculture, 3% industry and 4% background (ie geological origins) contributions. A high proportion of this (47ktonnesP/year) was determined to be soluble reactive P, that being immediately and wholly bioavailable. In the last decade tertiary treatments have been installed at larger WWTW making these UK average values lower in terms of sewage-derived overall contributions. However, considerable variation in sources exists between catchments, especially smaller ones. For example, Greene et al (2011) using a source model found that point source P in seven Irish catchments (10-100 km2) comprised 90% of the P inputs in one sewage impacted catchment but noted that 80% of this was feasible to remove by tertiary treatment. In contrast, the other catchments had a variety of sources. It is important to note that many rural sources such as septic tanks, farmyard runoff, piped discharges from roads etc, can behave like effluents in terms of their P delivery characteristics. The key factors for ecological damage are point sources that continue to discharge at times of low, summer river flow, when ecology is most sensitive. In contrast, many diffuse sources are rainstorm-driven and occur at high flow.

2.2.4. Key environmental legislation for P impacts in soils and waters

There is no EU-level P-specific legislation for the protection of waters and soils from the effects of inefficient agro-practice, soil management and P use in food systems. Policies indirectly addressing P can be roughly divided into three groups: water protection; agricultural management; waste management and soil protection and resource protection. The dominant EU-level legislation in each group is presented in Appendix III.1, Tables III.1a-d. These high-level legislative frameworks have been transposed into Scottish regulations (See Appendix III.2) and policies[3], and are outlined below.

Water protection

Regulations

- The Water Environment and Water Services (Scotland) Act 2003 (as amended) reported as the WEWS Act, which transposes the Water Framework Directive (WFD) (2000/60/EC) to national law.
- The Water Environment (Controlled Activities) (Scotland) Regulations 2011 (as amended), also known as CAR, which was developed to help SEPA implement the WEWS Act and support the RBMP process.

3 Regulations are used to impose and enforce minimum requirements for environmental quality. A policy refers to measures that are designed by the government and regulatory organisations to prevent or reduce harmful effects of human activities on the environment.

Policies

- SEPA's priority catchment approach agreed in partnership with the Diffuse Pollution Management Advisory Group (DPMAG) (DPMAG-SEPA 2017).

Agricultural management

Legislation

- The Nitrate Directive and the measures underpinning the implementation of the CAP, which do not explicitly address P.

Policies

- The Prevention of Environmental Pollution from Agricultural Activity (PEPFAA) (Scottish Executive 2005a).
- The Four Point Plan (Scottish Executive 2005b).

Waste management interactions with soil and agricultural protection

Regulations

- The Waste Management Licensing (Scotland) Regulations 2011, which includes Paragraph 7 exemption for the application of wastes on agricultural land following physiochemical analyses (See Section Appendix III.2) and
- The Animal By-Products (Miscellaneous Amendments) (Scotland) Regulations 2015 and the Animal By-Products (Enforcement) (Scotland) Regulations 2013.

Policies

- The British Standards Institution's Publicly Available Standards (PAS) for composts (BSI 2011; SEPA 2017a; WRAP 2016)) and AD digestates (BSI 2014; WRAP 2016).
- The Safe Sludge Matrix (ADAS 2011).

General resource management

Policies

- The Biosolids Assurance Scheme (BAS) Standard (UK BAS 2017).

2.3. The extent of P resources in recycled materials and issues in their use

2.3.1. Material mass budgets relevant to P resource recycling in Scotland

An analysis of P flows at a UK scale (Cooper and Carliell-Marquet, 2013) showed, at that time, a UK national P input of 138 ktP/year compared to 24 ktP/year being exported in industrial and food products; the inference being the difference was accumulating in the UK environment. Of this, 55 ktP/year was modelled to be entering centralised wastewater treatment works (WWTW), whereby 43% of that was discharged to waters. The remaining 32 ktP/year comprised WWTW sludges that were 70% recycled to land. However, the study highlighted that whilst sludge to land accounted

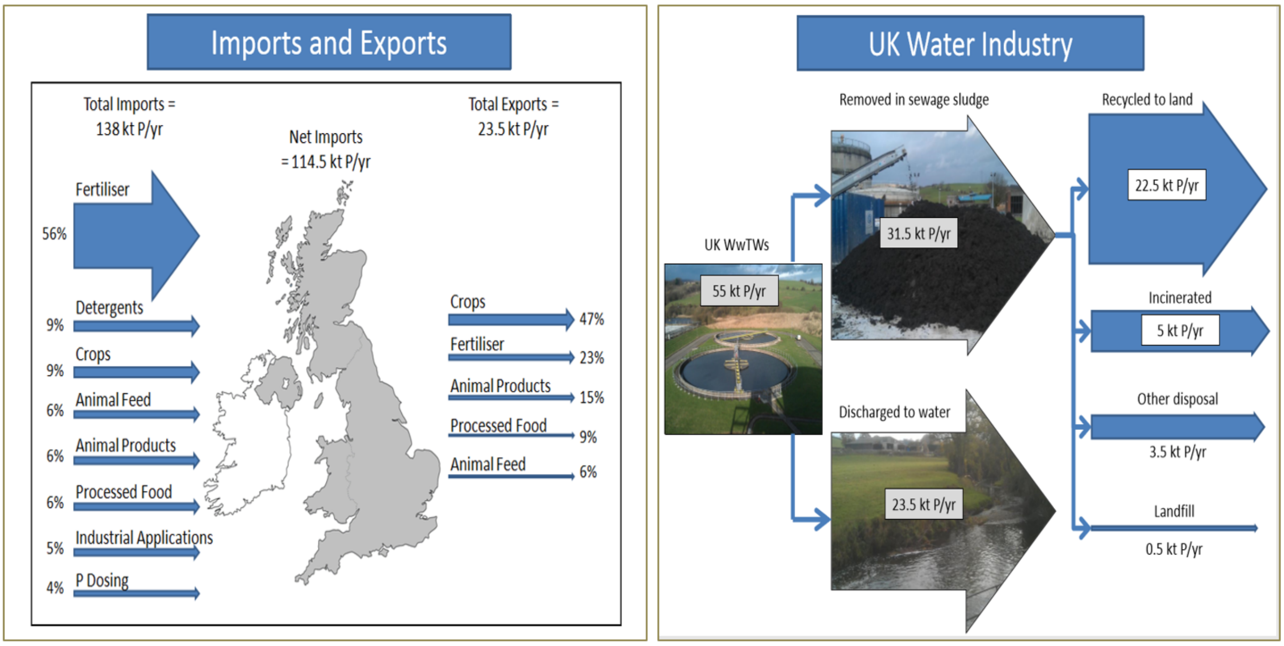


Figure 2. Illustrations of outputs from the UK-scale modelling study of P flows (Cooper and Carliell-Marquet 2013) using slides taken from Cooper and Carliell-Marquet (2012). In this study the fate of P in incinerated wastes were not examined in detail.

for 8% of the UK agricultural P requirement, it was applied to only 1.5% of the land area. Since previously sewage sludge application rates were calculated on the basis of crop N requirements regular over-application of P often occurred. Hence, P resources were not being used in an efficient way across wider agricultural areas and some smaller areas of application may have had diffuse pollution issues associated with losses of excess P applied. The spatial disconnect highlighted in the UK study by Cooper and Carliell-Marquet (2013) was a motivation for the present study to look at the connection between case-studies of centralised nodes of handling for P-rich materials, the catchment water pollution status and the opportunities and constraints of the local land bank.

The P resource flows budget specific to 2015 data sources for Scotland (Hough et al., 2016) has been presented in Box 1 (Section 2.1.2). The focus of this analysis was the internal P cycling processes that cascade from the major inputs such as PR-derived fertilisers, imported food and feed. Inputs and exports have not been quantified exhaustively. The study evaluated agri-environmental and water systems areas, looking at both P mass flows and their concentrations. The former highlighted the main areas of resource flows, whilst the latter show concentrated flows, which respectively are easier or more difficult for P recovery. This Scotland-specific analysis showed that P mass mainly existed in the agri-food sector, less so in wastes (these are in WWTW flows mainly). This situation in 2015 showed accumulations of ~10 ktP/year in soils and ~5 ktP/year in waters (in combination approximating to ¾ of Scotland's imported PR). The difficulties of recycling were highlighted since away from the agri-food sector the mass flows become numerous and smaller and potential recycling opportunities become dispersed and diluted in P concentration. In terms of P recycling, the total available waste-P stream accounted for could replace up to 40% of the 20 ktP/year chemical fertiliser inputs to agriculture; however, at the time, only 1.3 ktP/year of sewage, 0.3 ktP/year of AD and an unknown assumed small mass of compost, respectively, was being used on agricultural land. In terms of pollution to the water environment, the calculated sources of P losses to surface waters were 1.9 ktP/year from agricultural land and 2.4 ktP/year from water waste streams (a source distribution of 44% diffuse pollution from land, 53% from WWTW effluents and 3% from domestic septic tanks).

2.3.2. Sources of alternative P fertiliser materials

For a detailed evaluation of P inventories in the agricultural, food and waste sectors at a national level we refer readers to the datasets and their origins in Hough et al. (2016). A summary of the source material and P recovery processes to produce recycled-P fertilisers is given in Table 2. Appendix IV summarises the fertiliser requirements of different crops and what forms are currently being applied in Scotland, and more specifically the existing recycling practice associated with utilisation of animal-derived manures and slurries, the potential recycled-P resource from sewage and the potential for recycled-P resources as by-products from anaerobic digestion._

2.3.3. The fertiliser value of recycled-P materials

A general overview of the P or P_2O_5 content and solubility of chemical phosphate fertilisers and recycled-P fertilisers is presented in Appendix V.1 and quantitative evidence is presented in Table V.1. General properties of P fertilisers are summarised in Appendix V.2.

P solubility and crop P availability

P water solubility in fertilisers is often used as quality indicator for plant P availability (e.g. RB209). Other indicators of plant availability include: (i) the solubility in 2% citric acid (e.g. for assessment of the availability of Thomas phosphates, i.e. calcium phosphate); (ii) neutral ammonium citrate (e.g. for assessment of water-soluble fertilisers like superphosphate and triple superphosphate); and (iii) 2% formic acid extractable P (e.g. for assessment of PR). The composition of recycled-P fertilisers from waste is largely determined by the source material and process used for P recovery.

Chemical (PR-derived) fertilisers

Phosphate compounds found in unprocessed PR are not easily available to plants, for example PR water solubility has been found to be very low for all the phosphates, and thus water solubility values are not useful in selecting PR types (Zapata and Roy 2004). PR solubility can be increased, for example, by mixing it with sulphur and by co-composting it with organic matter (Korzeniowska et al., 2013 and literature cited therein).

Manure - Slurry

- Water soluble P in manures is approximately 15 to 31 percent of total P (Appendix V.1, Table V.1). Since water extracts include mainly primary and secondary alkali phosphates, primary alkali phosphates and their respective ammonia salts, Kratz et al. (2019) assumed that the organic P forms present in manures are quickly mineralised into these water-soluble forms. Despite a variety of manure compositions (e.g. animal types, diets, storage conditions) it is generally accepted in agronomic planning (see e.g. ADAS RB209 guidance) that 50% of the P is available to the following crop after application to soils.

Table 2. Source material and P recovery processes to produce recycled-P fertilisers (taken from Kratz et al., 2019; Möller et al., 2018; SEPA 2017a,b; Ford et al., 2017; Cieślik and Konieczka 2017; Schoumans et al., 2015; Zero Waste Scotland 2017).

Fertiliser type	Source material	P recovery process
Compost • Green waste (biomass) • Municipal waste	Plant residues of parks, gardens etc. /Urban household waste	Biological decomposition and stabilisation of biodegradable waste under controlled aerobic conditions (with oxygen) that results in a stable, sanitised material that can be applied to land for the benefit of agriculture, horticulture or ecological improvement.
Slaughterhouse waste and aka animal by products (ABP) or Bone / Meat meal	Slaughterhouse	Dry material produced after pasteurisation and/or sterilisation
Anaerobic digestate (AD) from • Household waste digestate • Industrial organic waste[4] • Farmyard manure • Dairy manure • Distillery waste	Urban household waste/ Catering and retailer organic waste/ Distillery waste used as feedstock to AD/farmland	Biological breakdown of organic matter into a gas (biogas), water and residual matter, which is made up of undigested material and dead micro-organisms. The mixture of water and residual matter is known as digestate. The process takes place in the absence of oxygen (hence it is anaerobic) and usually in sealed tanks known as fermenters or digesters.
Biosolids (biologically precipitated P)	Wastewater treatment works (WWTW) derived biosolids	Dewatered sludge whereby P bound in solid particles is biologically precipitated in the form polyphosphate granules (via enhanced biological phosphorus removal- EBPR*- from wastewater).
Biosolids (chemically precipitated P)		Dewatered sludge whereby P is chemically precipitated with addition of Al- or Fe-salts or Ca-oxide
Struvite		Precipitation of struvite in a phosphate rich stream resulting from EBPR.
Ashes • Sewage sludge • ABP ash • Wood ash • Cereal ash	WWTW/Slaughterhouse/farmland	Incineration producing P-containing ashes and subsequent treatment to enhance P recovery via Acidic digestion or leaching to transform P into water soluble forms Thermochemical treatment (mainly used in Germany), to produce calcium-silico-phosphate or calcium sodium phosphate.
Biochar • Sewage sludge: EBPR* or chemically precipitated • AD products • ABM • Softwood	WWTW/AD facilities/ Slaughterhouse/farmland	Pyrolysis (thermal treatment in the absence of oxygen) resulting in the production of calcium phosphates, and with increasing temperatures leading to an increasing degree of crystallinity. Depending on the additives used in the treatment, other P species than pure calcium phosphates may also be formed. Furthermore, inorganic pyro(- and poly)phosphates may be formed from orthophosphates by condensation/polymerization reactions during high temperature treatment. Pyrolysis also produces hydrogen (H_2), carbon monoxide (CO), carbon dioxide (CO_2), Methane (CH_4), ethane (C_2H_6), ethylene (C_2H_4), water steam and tars and a carbon-rich char.

*EBPR: Enhanced biological phosphorus removal, a sewage treatment configuration applied to activated sludge systems for the removal of phosphate.

4 In Scotland, a substantial source of AD is distillery waste (C. Erbert 2021, pers. com., June 23).

- Manures also show high solubility in neutral ammonium citrate and citric acid (Kratz et al., 2019).

Further information on the factors influencing P content and solubility in animal manures and slurries is given in Appendix V.1.

Recycled fertilisers (composts, anaerobic digestates, biosolids, struvite, ashes, biochars)

- Composts, digestates, ABP, biosolids and ash-based P fertilisers have a low water solubility for P, with less than 20% of P being water soluble (Appendix V.1, Table V.1).

P-fertiliser effectiveness

Few studies refer to comparative evaluation of the P (water) availability in inorganic and organic P fertilisers, i.e. their effectiveness as P fertilisers[5]. The available evidence shows that:

- Composts and biologically precipitated biosolids display higher effectiveness than chemically precipitated biosolids and ABP, presumably due to the higher content in Fe and Al phosphates in these biosolids and the apatitic structure of bone P in ABP (Möller et al., 2018 and literature cited therein). It is useful to note that the effectiveness of biosolids and ABP depends on soil pH and decreases with increasing pH and especially in soil pH>6.
- The three different forms of digestate, whole digestate, solid digestate, and liquid digestate, have various qualities, which dictate their utilizations and post-digestion treatment techniques (e.g. incineration, or struvite precipitation), with digestate quality mainly depending on the feed types and operating conditions of a specific anaerobic digester (Peng and Pivato 2019). For example, the P_2O_5/K_2O ratios of whole digestate from municipal solid and food waste is around 1:3, which is ideal for grain and rape seed growth. Möller et al. (2018) ranked the effectiveness of digestates from municipal feedstocks as P fertilisers between the highly effective composts and biologically precipitated biosolids and the less effective chemically precipitated biosolids and PR.
- Struvite has high effectiveness and thus from an agronomic point of view is one of the most desirable recovered P products not only due to the higher plant availability of the P forms it contains but also because its effectiveness is independent of soil pH (Möller et al., 2018). In addition, it has a low water solubility, which protects struvite-P from being adsorbed on soil colloids, and a high reactivity (Achat et al., 2014; Bonvin et al., 2015). It is also interesting to note that struvite has a slower dissolution rate when root growth is absent (Achat et al., 2014). However, there is a large variation in the effectiveness figures of struvite, potentially because of different crystal size of struvite from different source materials, which influences biogeochemical processes in struvite-amended soils (Möller et al., 2018).
- P availability in untreated ashes of any kind of organic matter is very low and strongly dependent on soil pH (Möller et al., 2018). Effectiveness depends on the feedstocks used and conditions during combustion (Nanzer et al., 2014), e.g. sewage sludge incineration ashes contain different amounts of sparingly available P -forms which reduce plant P availability. From an agronomic point view, any incineration approach represents a downgrading of the treated material in terms of specific P fertiliser value and the flows of N and organic matter. Ashes must be treated (e.g. acidic solubilisation) prior to application to slightly acidic, neutral or alkaline soils in order to get a plant available P fertiliser.
- P availability in chars depends on feedstock material and is inversely related to treatment temperature (Möller et al., 2018).
- Measurement of the relative P fertiliser effectiveness (i.e. the water soluble P fertiliser corrected for the P offtake from untreated control) shows that effectiveness increases from a minimum in ABP ashes to intermediate levels for PR, chemically processed sewage sludge, untreated ashes, urban digestates and biochar to high levels for animal manures, biologically processed biosolids, struvite and urban composts (Möller et al., 2018). This evidence shows that if a farmer needs external P inputs then manures, struvite and municipal composts should be preferred over PR.

2.3.4. Issues with contaminants associated with chemical fertiliser and comparisons with recycled-P materials

The composition of potentially toxic elements (PTE) in chemical-P and recycled-P fertilisers varies considerably for all parameters such as heavy metal, organic contaminants, and pharmaceuticals but also for pathogens. Available evidence on heavy metals and organic pollutants is reviewed in Appendix V.3. This evidence and any available evidence on other contaminants is summarised below.

Summary of material contaminant compositions

Emerging contaminants such as:

Heavy metals (Appendix V.3): Some elements especially cadmium (Cd) can be high in PR-derived fertilisers but this depends on the geographic and geologic origin of the ores used (Csillag et al., 2006). Low heavy metals

5 P fertiliser effectiveness: relative to a water-soluble mineral P-fertiliser (e.g. TSP), corrected of the P offtake from an untreated control (Moller et al., 2018).

concentrations have generally been found in composts, digestates and ABP from slaughterhouse wastes. However, the range of values in composts and digestates indicates the potential of exceedances of standards in the case of mercury and copper. Biosolids are characterised by the highest levels. Varying PTE loads were reported depending on the different approaches for production of sewage ash (Egle et al., 2013 cited in Möller et al., 2018).

Pathogens[6]*:* (in addition to manure and biosolids) these may be present in food waste, green waste household waste and digestate from infected animal products but also vegetables fertilized with contaminated organic amendments or irrigated with faecal contaminated water as well as from wildlife and rodents (Bloem et al., 2017). Factors such as reactor configuration, microbial competition, pH within the process and chemical interactions, e.g. concentration of free ammonia, contribute to the reduction of pathogens during anaerobic digestion (Ottoson et al., 2008). Pasteurization as a pre-treatment renders an organic amendment safe for recycling in terms of the most relevant zoonotic pathogens but apart from spore-forming bacteria (Bloem et al., 2017). Thermo-chemical treatment produces ashes low in trace elements without pathogens (Bloem et al., 2017). In summary, the pathogen content in biosolids and PAS110 digestate/PAS100 compost used on farmland in Scotland should be very low, due to the treatment being carried out to meet requirements of the accreditation standards.

Persistent Organic Pollutants (POPs) and Emerging contaminants (see also Appendix V.3): Biosolids contain the highest concentrations reported in the literature for recycled-P fertilisers, ranging from a few µg/kg (Weissengruber et al., 2018) to several g/kg DM (Möller et al., 2018 and literature cited therein; Kupper et al., 2008 cited in Möller et al., 2018). Urban waste composts are reported to have higher concentrations than ABP, digestates and struvite (Möller et al., 2018), but one study found POP concentrations in composts equal to background soil levels (Erhart and Hart 2010). The ratio of pollutants to P is important when pollutants inputs with P fertilisers are calculated. For example, polycyclic aromatic hydrocarbons (PAH) per unit P is higher in composts than in biosolids due to the low P concentrations of composts (Kupper 2008 cited in Möller et al., 2018).

Emerging contaminants:

- *Pharmaceuticals*: ABP may contain tetracycline (Kuhne et al., 2000). Very small and removable (through rinsing the crystals after filtration) concentrations are reported for a few studies on pharmaceutical content of struvite (Steinmetz and Meyer 2014 cited in Möller et al., 2018; Ronteltap et al., 2007). Manure, sewage sludge and effluents from sewage treatment plants can contain considerable amounts of antibiotics. Antibiotics are not reduced by the anaerobic digestion process.
- Endocrine disruptors: Per- and Polyfluoroalkyl substances (PFAS), are considered to be persistent for centuries and have been found in leachate from landfills, in biosolids and in the effluent from WwTW (see review by Akoumianaki and Coull 2019).

6 Pathogen content in biosolids and PAS110 digestate/PAS100 compost used on farmland in Scotland should be very low, due to treatment being carried out to meet standard requirements (A. Cundill 2021, pers. com. June 21).

2.3.5. Issues with applying the materials to land

A balanced assessment of P fertilisers is essential to account for a broad range of aspects from composition and plant availability to long-terms impacts on the environment (soil, water, atmosphere) and health. Here we review evidence on soil PTE accumulation risk, potential losses to water, GHG emission aspects (in production and utilisation) and public health effects.

Nutrient leaching risks from soils to waters

Aspects of the differences in the potential water pollution risks from different material applications to fields can be derived from comparing material P solubility in water (see Appendix V.1). From this, highly soluble P compounds in processed chemical fertilisers (88-100% P solubility in water) contrast with low solubility in ash, struvite and biosolids (trace to 10% water P solubility), moderate solubility for AD (<19%), where only some manures approach high P solubility (pig FYM up to 87%). Hence, in terms of direct solubility to waters many waste materials pose low risks of P leaching without subsequent transformations in soil, such as alteration by pH of the soil matrix, or microbial alterations (e.g. mineralisation). However, there is a considerable context-specific element to this based on soil compositions as well as the application methods (e.g. broadcasting onto the surface, ploughing in, injecting of liquid fractions) and the nature of the environment for pollution risks (leaching vs surface runoff and erosion). This is an area with a considerable need for research development as there are constantly new materials, blending of materials and variability in the feedstocks and end products of some waste streams that require wide ranging testing in different landscape risk situations. For now, results have to be inferred for the field situation from laboratory testing with new materials and older data from manures and sewage sludge trials.

Stutter (2015) examined a range of potential P replacement materials for their compositions and properties relating to crop P availability and leaching risks. The study showed that there were substantial differences in the P content and environmental (P leaching) behaviours of different waste materials when mixed with soil (in the case of this study a single Scottish soil that was quite strongly P fixing) and it must be understood how these can differ from chemical P fertiliser properties. A summary of data from Stutter (2015) is given in Table 3. High P densities were found in sewage sludge and digestate when dry matter contents were considered (although both had large moisture contents in raw material forms). Then four of the materials were studied for P availability and leaching risk where the assumptions were that a slow-release fertiliser where P became soluble slowly over time and the material did not migrate rapidly down the soil profile (i.e. leach to surface or ground waters) was good. The tested digestate material and pelletised chicken manure were found to have highly soluble P in water extracts and in the case of digestate there was a low degree of P sorption on the strong P retaining soil such that digestate seemed to have risk of soluble P migration down soil profiles and potential transport to waters if not used with appropriate amounts and timing to crop root uptake.

The study concluded that more research was needed in terms of three key questions:

- For a given P application what are relative crop P availability vs leaching risks?
- Are soil P tests appropriate for alternative P source materials?
- What are timescales for mineralisation of materials in soils (as these are very approximately derived in RB209).

The context specific nature of soluble P release and transport in a given landscape makes the quantification of differences in P leaching between an alternative fertiliser material and the chemical fertiliser that is being replaced very difficult. We are not aware of generalisations in models for such. In contrast diffuse pollution by erosion means is simpler since if erosion occurs then the P losses from the field are likely to be similar regardless of the type of solid fertiliser if the majority of that remained in the topsoil and would vary with the total P fertiliser usage, not fertiliser type. Erosion P losses would be different when comparing solid and liquid fertiliser products.

Soil PTE accumulation risk

This section summarises some of the risks inherent to the materials themselves. The actual risks in-situ depend on the nature of the site (soils, climate, landforms, transport distances to watercourses) and the agronomic circumstances (crop requirements and growth stage). Soil risk factors for planning are examined more in Section 2.3.6 (Box 2).

Risk assessment on the use of recycled-P in agriculture as part of "Zero Waste" strategies have also been carried out, usually based on mass balance modelling, with a view to predicting long-term soil accumulation of PTE such as heavy metals and persistent organic pollutants (POPs) (Amlinger et al. 2004; Smolders 2013; Möller et al., 2018; Weissengruber et al., 2018; Eriksen et al., 2009). Overall, there is a consensus about the soil PTE accumulation risk among studies. To illustrate, we summarise the findings presented by Möller et al., 2018 and Weissengruber et al., 2018, who accounted for soil background concentrations of potentially toxic elements (PTE) mainly heavy metals and persistent organic pollutants (POP) such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and dioxins/furans (PCDD/F, their public health-related limits and their inputs with fertilisers (chemical and recycled), atmospheric deposition and liming to predict change in soil concentrations over a period of 200 years.

Table 3. Summary results taken from Stutter (2015) on the compositions of seven potential fertiliser P replacement materials and P leaching risks for a subset of four materials. Materials were sourced within Scotland.

What material?		Basic characterisation				P leaching risk		P availability
Sample	Process and material sources	Moisture content ($g\ kg^{-1}$)	N ($g\ kg^{-1}$)	C ($g\ kg^{-1}$)	P ($g\ kg^{-1}$)	P water extraction (1g:40mL)	Sorption to strongly P fixing soil	Desorption to water after soil binding
Sewage sludge	Cambi-process, domestic sewage	72	39.6	321	20.8	1 mg/L	90	<1%
AD digestate	Slurry and abattoir waste	95	43.4	366	14.7	20 mg/L	4	5%
Green compost	PAS100 certified green waste source	40	13.5	381	2.7	2 mg/L	69	<<1%
Premium compost	Green plus food waste	52	19.1	257	3.1			
Chicken manure	Commercial dried, pellitised	<5	40.4	335	6.2	28 mg/L	28	<<1%
Biochar	Softwood pallets	<5	1.4	573	0.2			
Seaweed	Washed in freshwater, air-dried	80	26.9	367	2.8			

The mass balance modelling[7] applied in these studies predicted (Möller et al., 2018; Weissenburger et al., 2018):

1. A low to moderate risk of soil quality degradation due to PTE inputs with recycled-P fertilisers.
2. A key influence of soil pH and precipitation on PTE leaching from soils to water systems, with a relatively low precipitation excess (e.g. 100mm) at pH=7 increasing the leaching risk relative to soil background PTE concentrations and threshold values set by legislation (as reported by Möller et al., 2018).
3. A lack of correlation between heavy metal concentration in recycled-P fertilisers and in soil, with fertilisers having a low to intermediate heavy metal content such as green waste and urban organic waste composts posing a higher heavy metal accumulation risk in soil after long-term application than biosolids and struvite, which have a low heavy metal: P ratio.
4. Efficient reduction of soil PTE accumulation risk associated with the use of sophisticated techniques to recover P from sewage like struvite precipitation and thermal approaches.
5. Triple superphosphate (TSP), PR, Green waste compost and organic household waste compost have the highest predicted soil accumulation risk for Cadmium.
6. PR, Green waste compost and organic household waste compost have the highest Heavy metal to phosphorus index.
7. PR has the highest heavy metal to total combined nutrient value index.
8. Very low soil heavy metal (except for Cd) accumulation risk from the use of PR-derived fertilisers (PR, TSP).
9. Similar PCB inputs with recycled-P fertilisers to atmospheric deposition, and no effect of contamination level (high or low) of PCB in recycled-P fertilizers on soil PCB accumulation risk.
10. Higher soil PAH accumulation risk for composts compared to other fertilizers and atmospheric deposition.
11. Negligible soil PCDD/F accumulation risk.

7 A mass balance model based on Smolders (2013) was used by Möller et al., 2018 to assess the accumulation of heavy metals in soil. The mass balance approach applied by Smolders (2013), Six and Smolders (2014) and the European Scientific Committee on Toxicology, Ecotoxicology and the Environment, CSTEE (2002) for assessing the cadmium accumulation in European agricultural soils was used by Weissengruber et al., 2018. A mass balance model according to Amlinger et al. (2004) was used by Weissenburger et al., 2018 to assess the accumulation of persistent organic pollutants. Weissengruber et al., 2018 used soil background levels taken from an European survey (Salminen et al.2005).

To sum up:

- Mass balance modelling showed that the ratio of PTE to P concentrations are the main driving factor influencing PTE flows applied to the soil (Möller et al., 2018; Weissenburger et al., 2018). The reason is that recycled application rates are determined by P (generally nutrient) concentrations so that materials with both low PTE and P content can result in a higher risk of PTE accumulation in the soil. As a result, fertilisers having a low to intermediate PTE content such as green waste and urban organic waste composts pose a higher PTE accumulation risk in soil after long-term application than biosolids and struvite, which have a low PTE: P ratio.
- The finding of low soil PTE accumulation risk from biosolids is also associated with the continuous reduction in the inputs of toxic elements through waste streams such as sewage systems due to implementation of environmental regulations, including a stricter separation of sewage from industry and households, and changing waste management technologies (Eriksen et al., 2009; Schroder et al., 2008).
- The implementation of source segregated collection of urban organic household wastes has reduced the PTE concentration in composts typically by a factor of 2–10 compared to mechanically separated material (Amlinger et al. 2004).
- The implementation of the Aarhus Convention and the Stockholm Convention on POPs in combination with technical improvements in industry like filters and high temperature combustion (Leschber 2006) have also contributed to lower PTE to P ratios in recycled-P fertilisers, and therefore to reducing soil PTE accumulation risk.
- Heavy metal behaviour in the soil varies with pH. For example, Cu tends to bind to organic ligands and Pb is strongly absorbed by humic matter at pH 4 and above (Weissengruber et al., 2018) and leaching is initiated when the associated organic matter solubilises and is transported. On the other hand, leaching was most relevant for Cd, Cr, Ni and Zn at pH 5, with mass balance modelling predicting decrease of the risk of Cd, Ni and Zn accumulation in the topsoil with decreasing soil pH and increasing precipitation excess due to increasing leaching losses (Weissengruber et al., 2018 and literature cited therein).

2.3.5.1 Climate change risk: GHG emissions

Greenhouse gases[8] (GhG) emissions from the use of PR-derived and recycled-P fertilisers can be estimated from energy consumption in the following processes: P production (e.g. PR mining and fertilizer and animal feed production); P recycling processes (e.g. from manure, wastewater and composting); and transport of fertilisers. See Table 1 (Section 2.1.2) for a comparative presentation of GhG emissions from a wide range of P fertilisers.

Studying GhG emissions from P fertilisers is new research topic and current evidence is limited in Scotland and the UK. In Scotland, a CxC study found that GhG emissions arising from the storage and spreading of livestock manures[9] (slurry and solid manures) account for just under 2% of all GHG emissions and just under 10% of GHG emissions from agriculture in the country (Wiltshire, 2018). Approximately 92% of Scotland's GHG emissions from slurry management are from housing and storage, and approximately 8% are from field application (see footnote).

Studies outside of Scotland show variation in estimated GhG emissions from P recovery processes. Generally, larger GhG emissions are predicted globally from P recycling than mining due to higher energy demand, with P recovery from wastewater contributing by 70% to GhG emissions from all P recycling processes (Golroudbary et al. 2019 and literature cited therein). Anaerobic digestion of household wastes tends to emit more CH_4 than aerated home composting with decentralised (small-scale) aerobic composting, anaerobic digestion and vermicomposting bins, emitting less GhGs when compared with larger scale systems (Chan et al., 2011). However, it can be argued that some processes will take place anyway for other reasons (e.g. AD for energy generation) with P being a by-product. Additionally, anaerobic digestion of household/municipal waste has an advantage over composting, incineration or combination of digestion and composting, mainly because of better energy balance (Mata-Álvarez et al. 2000) and lower global warming potential in kg CO_2 eq. kg^{-1} P (; Möller etal., 2018). Further, biochar application to soil has been proposed as a means for reducing soil GhG emissions (Lehmann et al., 2006). The effects, however, depend on crop type as well as on soil properties, moisture and temperature (Fidel et al., 2019). More examples are presented below, but a thorough review of GhG emissions from recycled-P processes is beyond the scope of this report.

- Global average GhG emissions from conventional P fertilisers are 1.36 kg of CO_2 equivalents (eq) per kg of P_2O_5 produced by PR (Kool et al., 2012).
- Annual GhG emissions from biosolids stockpiles in fields depend on their age and range from 90.3 kg CO_2-e Mg^{-1} in one year old stockpiles to 27.5 CO_2-e Mg^{-1} in stockpiles older than three years (Majumder et al. 2014; 2015).
- GhG negative emissions (as soil carbon storage) from biosolid application to land vary from -580 kg CO2-e/t of dry solids from composted and lime stabilised sludge to -410 kg CO2-e/t of dry solids from thermally dried and digested cake to -370 kg CO_2-e/t of dry solids from liquid digested sludge (Thorman et al., 2009). However, biosolids addition to land can increase N_2O emissions.
- Net GhG negative emissions from struvite production at a P_2O_5 content of 25% are -1.40 kg CO_2 eq. per kg of P_2O_5 and can outweigh transport positive GhG emissions (de Vries et al. 2017).

Public health risk

When PAS accredited composts and digestates are used it has been proposed that there is no public health risk (Longhurst et al., 2019). However, pharmaceuticals, POP and other emerging contaminants have not been given thresholds and are not tested within the PAS scheme. In terms of pathogens, meat and bone meal can be considered as much safer than animal manure and several other organic fertilizers (Möller 2015). The risk to human health via dietary intake of heavy metals and POPs from crops grown on biosolid treated soils was found to be minimal (Eriksen et al. 2009; Haynes et al. 2009; Smith 2009; Stutt et al., 2019).

Low concentrations of antibiotics and their metabolites enter the food chain when plants are grown on fields which received organic nutrient sources contaminated by antibiotics. Despite the low concentrations these pose environmental and direct and indirect human health risks as repeated exposure to low concentrations of antibiotics causes genetic variation and results in the transfer of Antimicrobial Resistance Genes (ARGs) and finally the generation of resistant pathogens and bacteria (Bengtsson-Palme et al., 2018; Bloem et al., 2017). In addition, resistant bacteria enter the soil directly via the application of organic nutrient sources and ARGs may be transferred to indigenous environmental bacteria by horizontal gene transfer (Bloem et al., 2017).

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8 Mainly Carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4).

9 Methane nitrous oxide emissions from manure management were calculated following IPCC methodology. Emissions of methane following application of slurry to land are negligible (IPCC, 2006, paragraph 10.4) and do not appear to be calculated. Indirect emissions of nitrous oxide from leaching and runoff following slurry application to land were estimated according the 2006 IPCC guidelines using equation 11.10 and the default nitrogen leaching/runoff factor. The fraction of nitrogen that is leached is a country specific value.

2.3.6. Control measures for pollution in applying materials to land

Accreditation of materials has developed to provide a judgement of a material's composition with respect to how it should be regulated. The main accreditation schemes and their standards applicable here are PAS 100 (BSI 2011) for composts (Table 4) and PAS110 (BSI 2014) for digestates (Table 5). However, it should be noted that the target limit for the specified contaminants should be zero, to near zero and the values represent upper permissible limits.

In terms of how this affects the material usage, BSI certified composts PAS 100 (BSI 2011) and digestates PAS110 (BSI 2014) and, can be applied to land without further regulation, providing they are used within the terms of SEPA's guidance on "Regulation of Outputs from Composting Processes". However, we have no data on how much of the compost and digestate spread on land in Scotland meets the BSI PAS criteria. The AD that is produced on-farm from farm-derived wastes can be spread to farmland without requiring PAS110 accreditation or paragraph 7 exemption. There is no stipulation that this farm-produced material must be applied on the farm of origin either (see: SEPA 2017b). Composts and digestates from processes which do not comply with the BSI PAS criteria, or the farm use noted above will otherwise be subject to full waste regulatory controls by SEPA, e.g. Paragraph 7 exemption requirements of The Waste Management Licensing (Scotland) Regulations 2011 (Appendix III.2, Box III.2.1). The only available source of data about the nutrient and PTE content of composts and digestates applied to land under Paragraph 7 exemption is a review by SEPA, where Cundill et al. (2012) presented the nutrient and PTE content of composts. More recently, Stutt et al. (2019) assessed organic contaminants in compost as unlikely to be present at levels posing risk to human health or the wider environment.

In the planning and regulation of applications of biosolids sludge to land there are interactions with the specific site conditions in terms of both the condition of the soil receiving the materials and the type of crops and their associated risk factors. Figure 3 shows an example of the UK wide guidance of The Safe Sludge Matrix (ADAS 2001) in terms of the suitability of applications of different levels of sludge treatment and allowances for application to different crop types. Table III.2.1 in Appendix III.2, gives maximum (pre-addition) permissible levels of PTEs (in this case trace metals) in soil for considering application of sewage sludge under the Sludge (Use in Agriculture) Regulations 1989 (as amended), which applies in England, Wales, Northern Ireland and Scotland. In Scotland spreading on non-agricultural land, non-food crops and for ecological benefits such as land reclamation is regulated instead by Paragraph 8: Exemptions (The Waste Management Licensing (Scotland) (WMLR) Regulations 2011; SEPA 2020). The Sludge (Use in Agriculture) Regulations 1989 recognise the pH-dependency of availability of the metals zinc, copper and nickel and set more stringent requirements for more acidic soils. All applications are prohibited for soils pH <5.0. Soil risk factors are shown in Box 2 (Section 2.3.6).

Table 4. British Standards Institution (BSI) PAS 100 Safety-related parameters and limits in composts. Source: BSI 2011; SEPA 2017a. At the bottom are some updated values for plastic contaminants.

Contaminant parameters	Reported as (units of measure)	Limits
Chemical		
Cadmium (Cd)	mg/kg (ppm) dry matter	≤ 1.5
Copper (Cu)	mg/kg (ppm) dry matter	≤ 200
Chromium (Cr)	mg/kg (ppm) dry matter	≤ 100
Lead (Pb)	mg/kg (ppm) dry matter	≤ 200
Nickel (Ni)	mg/kg (ppm) dry matter	≤ 50
Mercury (Hg)	mg/kg (ppm) dry matter	≤ 1
Zinc (Zn)	mg/kg (ppm) dry matter	≤ 400
Biological		
Salmonella spp.	MPN / 25 g	Absent
Escherichia coli	CFU g^{-1}	≤ 1000 CFU g^{-1}
Weed seeds	Viable propagules / litre	≤ 5 maximum*
Phytotoxicity	Score % of control	80% minimum
Physical		
Total glass, metal and plastic of which plastic	% m/m air-dried sample > 2mm % m/m air-dried sample > 2mm	≤ 0.5* ≤ 0.25*
Stones and other consolidated mineral contaminants > 2mm	% m/m air-dried sample	≤ 7*

Date	Parameter	Limit
From 1 December 2018	Plastic	0.08% (66% of current PAS100)
From 1 December 2019	Plastic	0.06% (50% of current PAS100)

Table 5. Test parameters, upper limit values and declaration parameters for validation of digestates. NOTE: (1) PAS 110 (BSI 2014) does not require testing and declaration of all water-soluble nutrients and elements. (2) Total nitrogen is the limiting factor for PTE and physical contaminant contents. For example, a total nitrogen content of between 2 and 2.9kg/t means that Cd could not exceed 0.36mg/kg, and stones could not exceed 9.6kg/t. PTE: Potentially toxic elements; ABP: Animal by-product; WD/SL/SF: Whole Digestate/Separated Liquor/Separated Fibre; dp: decimal places. Source: BSI 2014.

Parameter	Method of test	Upper limit and unit
Pathogens (human and animal indicator species) in WD/SL/SF		
ABP digestate: human and animal pathogen indicator species	As per appropriate ABP regulation or any other method approved by the competent authority/Animal Health vet/Veterinary Service vet	As specified by the competent authority/Animal Health vet/Veterinary Service vet in the "approval in principal" or "full approval"
Non-ABP digestate: *E. coli*	SCA MSS Part 3A [N1] or BS ISO 16649-2	1,000 CFU/g fresh matter
Non-ABP digestate: *Salmonella spp.*	Method as specified by appropriate ABP regulation, according to nation in which digestate is produced, or SCA MSS Part 4A [N2]	Absent in 25 g fresh matter
Potentially toxic elements (PTE) in WD/SL/SF		
Liquid (≤ 15% TS) digestates	For all PTEs [4)]: BS EN ISO 15587-1:2002	Declare on a fresh weight basis
Fibre (> 15% TS) digestates	For all PTEs [4)] except Hg: BS EN 13650:2001 For Hg: BS ISO 16772	Declare on a fresh weight basis

Total nitrogen (N)	**kg/t**	**Less than 1**	**1 to 1.9**	**2 to 2.9**	**3 to 3.9**	**4 to 4.9**	**5 to 5.9**	**6 to 6.9**	**7 to 7.9**	**8 to 8.9**	**9 or more**
Cadmium (Cd)	mg/kg	0.12	0.24	0.36	0.48	0.60	0.72	0.84	0.96	1.08	1.2
Chromium (Cr)	mg/kg	8	16	24	32	40	48	56	64	72	80
Copper (Cu)	mg/kg	16	32	48	64	80	96	112	128	144	160
Mercury (Hg)	mg/kg	0.08	0.16	0.24	0.32	0.40	0.48	0.56	0.64	0.72	0.80
Nickel (Ni)	mg/kg	4	8	12	16	20	24	28	32	36	40
Lead (Pb)	mg/kg	16	32	48	64	80	96	112	128	144	160
Zinc (Zn)	mg/kg	32	64	96	128	160	192	224	256	288	320

Total nitrogen (N)	**kg/t**	**Less than 1**	**1 to 1.9**	**2 to 2.9**	**3 to 3.9**	**4 to 4.9**	**5 to 5.9**	**6 to 6.9**	**7 to 7.9**	**8 to 8.9**	**9 or more**
Total stones	kg/t	3.2	6.4	9.6	12.8	16	19.2	22.4	25.6	28.8	32
Total physical contaminants (excluding stones)	kg/t	0.04	0.07	0.11	0.14	0.18	0.22	0.25	0.29	0.32	0.36
No "sharps" (see Note to **6.1**)											

THE SAFE SLUDGE MATRIX

CROP GROUP	UNTREATED SLUDGES	CONVENTIONALLY TREATED SLUDGES	ENHANCED TREATED SLUDGES
FRUIT	✗	✗	✓ 10 month harvest interval applies
SALADS	✗	✗ (30 month harvest interval applies)	✓
VEGETABLES	✗	✗ (12 month harvest interval applies)	✓
HORTICULTURE	✗	✗	✓
COMBINABLE & ANIMAL FEED CROPS	✗	✓	✓
GRASS & FORAGE - GRAZED	✗	✗ (Deep injected or ploughed down only) 3 week no grazing and harvest interval applies	✓ 3 week no grazing and harvest interval applies
GRASS & FORAGE - HARVESTED	✗	✓ (No grazing in season of application)	✓

Figure 3. Provisions of The Safe Sludge Matrix (ADAS 2001). "Conventionally treated sludge" is sludge that has been subjected to defined treatment processes and standards that ensure at least 99% of pathogens have been destroyed. "Enhance treated sludge" is sludge that is free from *Salmonella* and has been treated so as to ensure that 99.9999% pathogens have been destroyed (a 6 log reduction).

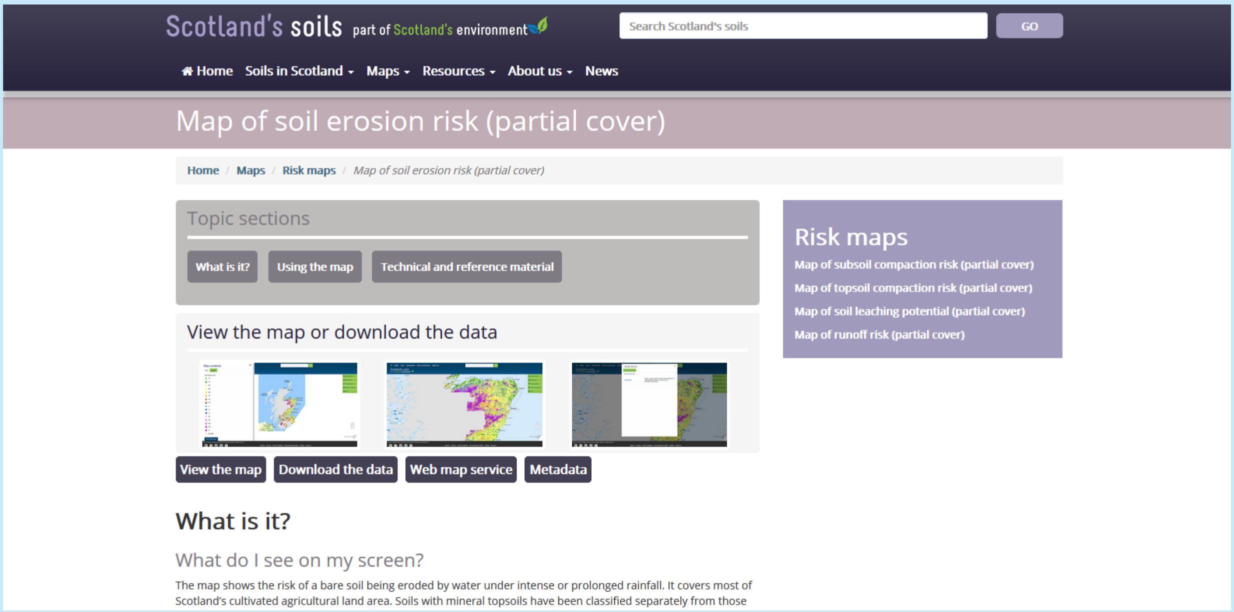
✓: All applications must comply with the Sludge (Use in Agriculture) Regulations 1989 (as amended).

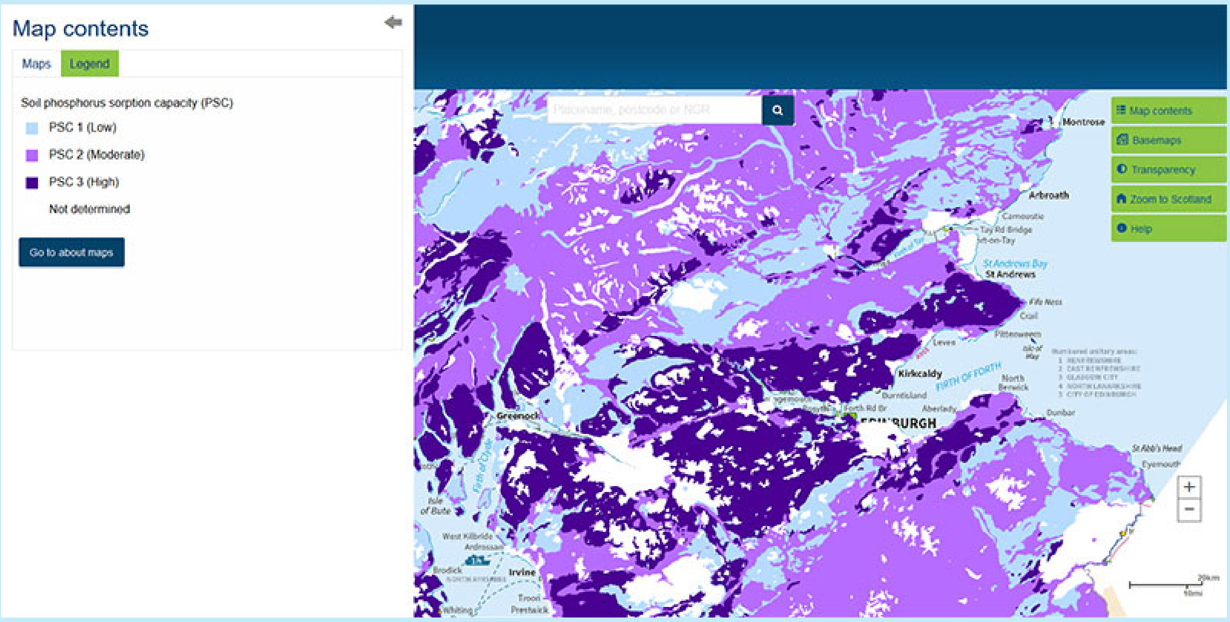
x: Applications not allowed (except where stated conditions apply)

Box 2. Planning tools to assess local context of risks from nutrient pollution using the national soils data and developed risk assessments.

Based on translation of national soil mapped data several risk assessments have been developed that inform national to field scale planning of nutrient usage and potential for pollution losses via erosion and leaching. These include:

- Soil erosion risk
- Soil leaching potential
- Soil phosphorus sorption capacity





2.4. Summary of the benefits and issues of potential recycled materials compared to chemical P fertiliser

Table 6 provides a qualitative appraisal of the key characteristics of chemical and recycled fertilisers based on the information reviewed in the previous sections[10]. The appraisal uses a comparative approach for the benefits and risks from the production and use of each type of fertiliser and the RAG (Red-Amber-Green) rating system. Thus, each type of fertiliser is compared to all other types of fertilisers for each impact category (see Section 2.1.2: LCA): red indicates higher risk for constraints and harmful or negative impacts and lower potential for benefits; amber indicates moderate benefits or moderate risks and constraints; and green indicates lower risk for constraints and harmful or negative impacts and greater potential for benefits.

Table 6 shows that most approaches to P recycling have benefits and risks:

- From a sustainable management perspective (see also Section 2.1.2), the advantage of using composted materials, anaerobic digestate, biosolids and untreated sewage sludge ashes derives from their lower potential for PR depletion, fossil fuel use for their production and P recovery rates[11] compared to chemical fertilisers and struvite. Manure and slurry present lower risk for resource and fossil fuel use but have a lower P recovery rate than all the other types of fertilisers.
- In terms of agronomic benefits (see also Section 2.3.3.2), manures/slurries, composts and (biologically precipitated) biosolids show the greatest relative P fertiliser effectiveness but this may decrease if the soil pH is outside of optimum ranges for nutrient and trace element availability. A key agronomic consideration is the overall nutrient value of the potential material, also considering N, K, Mg and sometimes beneficial trace elements. Often these wider elements are not in an optimum balance with P such that some components suffer application or balancing with other specialised fertilisers. Struvite effectiveness can be very high but varies with the crystal size of struvite from different source materials, which influences biogeochemical processes in struvite-amended soils. Effectiveness of chemical fertilisers (e.g. PR), chemically precipitated biosolids and biochars is relatively lower and depends on various factors including Fe and Al content, soil pH and, in the case of chars, on the feedstock used and treatment temperature. Anaerobic digestate has a higher effectiveness than PR but lower than composts. Untreated ashes have the lowest effectiveness due to negative changes in the material properties[12] during incineration.
- In terms of eutrophication risk (see also Sections 2.1.2), TSP, manures, composts and struvite pose the greatest risk. Digestates and biosolids pose the lowest risk. However, the P losses to watercourses from land amended with biosolids are poorly studied. Eutrophication risk from PR and untreated sewage sludge ashes can be ranked between high and low risk fertilisers. Further research is needed to elucidate leaching risk from the application of recycled-P fertiliser in the context of their effectiveness (see section 2.3.5.1).
- In terms of soil contamination and associated public health risks (see also Sections 2.3.5.2 and 2.3.5.4), chemical fertilisers, ashes and struvite pose the lowest risk for heavy metals, emerging contaminants, pharmaceuticals and microbes. However, there are caveats: PR application increases the risk from Cd contamination; risk from sewage sludge ashes depends on type of treatment, original material and contaminant and is generally poorly studied; and risk from contaminants in struvite has been poorly studied. Further, contamination risks from manure/slurry and biosolid application depends on management, soil type, weather, and various other contaminant-specific factors. In addition, for certain type of recycled P fertilisers, risk derives from particular types of contaminants such as: Hg and Cu in composts and digestates; high PTE:P ratio in sewage sludge ashes; pharmaceuticals in composts and digestates; and spore forming bacteria in composts.
- In terms of GhG emissions (see also Section 2.3.5.3), production of chemical fertilisers, digestates and struvite results in the lowest CO_2 emissions, which can compensate transport emissions (e.g. in the case of struvite). Manure/slurry production and application are associated with a very low contribution to Scotland's GhG emissions, but this is poorly studied on a global level and wider issues with atmospheric N pollution (e.g. ammonia losses) occur. For other P fertilisers, GhG emissions depend on scale (e.g. composts, with small-scale systems emitting less) and the type of gas emitted (e.g. higher risk from N_2O than from CO_2 associated with biosolid application).

10 The results of our appraisal shown in Table 6 were also checked against the Multicriteria Analysis scoring performed by Moller et al., 2018 for the same types of fertilisers and impacts/characteristics.

11 compared to P concentration in PR, as control for P recovery

12 Biosolid incineration downgrades the material from a soil fertility perspective by reducing the plant P availability, the recovery rates of other nutrients (Nitrogen and Sulphur), and the organic matter inputs to the soil (Möller et al., 2018).

The regulatory landscape is evolving for both chemical and recycled P fertilisers under geopolitical, environmental and public health-related concerns. (Table 6: Development of appropriate regulatory processes). Our review showed that the regulatory framework addresses agricultural management options related to application of PR and recycled fertilisers such as manure and slurry. There is also considerable progress in regulations enabling reduction of various waste streams but despite the development of policies (e.g. PAS 100/110) it is unclear how these translate to ensuring safe use of recycled-P fertilisers such as biosolids. For example, no standards have been established yet for emerging contaminants in composts, digestates and biosolids. Further, certain recycled fertilisers such as biochar, ashes and struvite are not mentioned in the regulations that are reviewed in Section 2.2.4.

To sum up, this review of the characteristics and properties of recycled P fertilisers shows that:

- There are many alternatives to fertilisers derived from PR; but the choice of recycled P fertiliser must strike a balance between suitability of a fertiliser (e.g. accounting for soil pH and the other nutrients present[13]) and risks associated with the potential of soil contamination, losses to water, crop/food contamination and GhG emissions.
- The highest benefits in terms of sustainability, agronomic effectiveness, environmental protection and compliance with the regulations can be achieved by rather simple approaches, such as manures, composts and biosolids, which can be locally available and have low transport cost. However, these approaches also have risks, e.g. accumulation of organic pollutants, which may affect soil fertility and food security.
- Any approach to producing and applying recycled P fertilisers has strengths as well as weaknesses; therefore, choice of recycled P fertiliser is always a balancing act between benefits and constraints.

13 Other nutrients, especially N, K and Mg also play a role but examining their role was outside of the scope of this report.

Table 6. Summary of relative benefits, practicalities and risks taken from the combined evidence of this review. Green: limited constraints/high benefits. Amber: moderate constraints/moderate benefits. Red: limited benefits or considerable constraints. Evaluation of impacts on resource depletion and fossil fuel use, and P-recovery are based on Table 1. Impacts of PR and TSP are shown separately for resource depletion, fossil fuel use, GhG emissions, eutrophication, P-recovery and overall recycling potential in Table 1. P recovery: compared to P concentration in PR, as control for P recovery; P fertiliser effectiveness: relative to a water-soluble mineral P-fertiliser (e.g. TSP), corrected of the P offtake from an untreated control.

Impacts	Chemical fertilisers (PR/ TSP)	Manure/slurry (raw)	Compost	Anaerobic digestate	Biosolids (dewatered sewage sludge)	Biochar	Untreated sewage sludge ashes	Struvite
Resource depletion	1							
Fossil fuel use	1					n.d.		
P recovery	Control							
P fertiliser effectiveness	Control							
Eutrophication risk	TSP PR				Poorly studied	n.d.		
Risk of heavy metals accumulation in soil	Cd		Hg, Cu	Hg, Cu			2 Low PTE:P	2
Risk of emerging contaminant accumulation in soils (antibiotics, PFAS, microplastics)			Antibiotics?	Antibiotics?		n.d.		
Potential of microbial contamination and alteration, including antimicrobial resistance genes proliferation			Spore-forming bacteria					
Relative risk of GhG emissions (note: all applications of N rich materials pose some threats)		Scotland ?	large systems small systems		CO_2 N_2O			
Development of appropriate regulatory processes		Spreading	3	3				

1. P concentration in PR is much higher and therefore the consumption of abiotic resources (e.g. inputs of chemicals) is lower per unit P recovered for PR than for TSP.

2. Depends on type of treatment, original material and contaminant but generally poorly studied.

3. No standards for pharmaceuticals/POPs; lack of data

? Limited evidence or poorly studied.

3. Selection of catchment case studies

The catchment screening was based on:

- Evaluation of national river catchment monitoring data to show rivers failing Water Framework Directive compliance based on soluble reactive P (SRP) mean annual concentration failures (ie highlighting those below 'good' status for P) over the last decade.
- Selection of those rivers that had consistent failing SRP concentrations, or declining quality that resulted in them failing in the latter part of the last decade.
- Restriction to rivers <250 km^2 catchment areas.
- Evaluation of dominant and secondary land cover using LCS88 datasets to demonstrate an agricultural land bank suitable for fertiliser replacement.
- An assessment of the number of licenced effluent discharge sites (CAR) (data from SEPA) looking to incorporate several within the catchment in order to supply recycled P and remove P from waste streams discharged to surface waters.
- An evaluation of the number of anaerobic digestion plants taken from the NNFCC website (https://www.biogas-info.co.uk/resources/biogas-map/) looking to include several of the catchments with AD as an additional recycled P source.

The project steering group was consulted for any prioritisation of catchments and decisions made on the criteria above with a goal to attain two catchments for the application of the P offsetting model.

The full catchment selection matrix (Appendix VI.1) shows the selection of river waterbodies where WFD monitoring showed failures specific to the annual mean concentrations of SRP for catchments of areas restricted to those nationally of size <250 km^2. From this selection two catchments were initially selected, but due to the amount of research involved in identifying material availability and land bank in the first catchment, only analysis of the first catchment was carried out.

- ***Bonny Water/Red Burn*** (SEPA waterbody ID 4205)
- Throughout 2007-2016 the river water quality with specific regard to soluble reactive P concentration remained poor to moderate (failure to attain Good Ecological Status). This catchment area contained one location in the CAR database of national monitored effluents (Dunnswood WWTW) and one anaerobic digestate plant. The dominant and secondary land cover types in the catchment were arable with intensive grassland, respectively. This became the dominant focus case study in this report.
- ***River Farg*** (SEPA waterbody ID 6701)
- This catchment, with one major WWTW, one large septic tank identified and an AD plant, was initially selected since water P status declined to moderate (failure to attain Good Ecological Status) from 2013. Analysis of River Farg did not take place.

4. Phosphorus offsetting model methods

The overall factors considered are summarised in Figure 4. Then the methods subsections described below follow the same order as 1-5 in the flow diagram.

4.1. Mass and form of recyclable P

4.1.1. Amounts and compositions of P bearing materials available from within-catchment and nearby surrounding sources

Biosolids from WWTW

Analysis of the main WWTW (Dunnswood) within the catchment area of the WFD waterbody was supplemented by analysis of materials from two further WWTW (Bonny bridge and Denny) that were situation very close to the outside of the catchment. The inventory of P mass balance and fates of sludges were partially supplied by Scottish Water. Taking P stocks and flows data we had an assumed sludge availability determined as: P resource inflow – final effluent outflow. To evaluate the raw sludge resource and handling constraints this P mass had to be understood in terms of a bulk sludge materials. For this purpose, we assumed the material from Dunnswood WWTW was available as digested cake and used reference values of 11 kgN/tonne FW and 18 kgP_2O_5/tonne FW taken from SRuC (2015). As a result, whilst the P mass available from WWTW in the area was accurately derived from actual WWTW-specific data, the fresh weight of material and hence handling and transport costs came from an average value in this proof-of-concept study. However, given the high output from anaerobic digestion compared to WWTW-derived P this was unlikely to affect the study conclusions. The summary of P masses, material forms and the information gained on material transport is reported in the results (section 5).

Hypothetical scenario on potential struvite recovery at Dunnswood WWTW

A struvite generation scenario was developed to provide a representation of the potential of a high-grade replacement fertiliser product material in terms of capability to be used on a wide range of crop types, high P density and easy handling. The potential struvite production was based on only the Dunnswood WWTW and used data from the Phosphaq and Ostara processes reported in Kleemann et al. (2015). In this scenario struvite removed P from effluent and sludge components at the WWTW. According to Kleeman et al. (2015) this reduced the P content of sludge cake from 1.17 to 1.07% w/w DM P content (~9% reduction). Whilst benefits include improved drying of sludge cake (lowering transport costs), it also lowers the sludge P content. However, it should be noted that sewage sludge is generally too high in P compared to the N need of crops, and hence, reducing the P content means more sludge can be applied and therewith more of the required N (whilst this study focusses on P sustainability that of N is also important since artificial N fertiliser is costly and very energy intensive).

Anaerobic digestate

The resources available from the commercially-run food waste feedstock anaerobic digestion plant at Cumbernauld were derived from values on the companies webpages (https://www.energenbiogas.co.uk/ and https://www.energro.co.uk/) and confirmed by a phone call.

Manures and slurry

The amount of manure and slurry was calculated as the area-specific stocking density multiplied by grassland areas multiplied by a literature value for daily amounts produced by housed animals. An average stocking rate for the catchment was derived from Matthews et al. (2012) using data from the June agricultural census and December survey for 2009. Their method determined total livestock units (LU) as cattle (aged one year and over) number multiplied by a factor of 1 and sheep numbers multiplied by a factor of 0.12 (farmed deer were considered negligible for this area). For the Bonny Water WFD waterbody the pixels in their mapping showed ranges of 0.51 to 1.00 and 1.01 to 2.00 LU / ha across the land cover types of common grazing, rough grazing, grass over 5 years, grass under 5 years. For the current modelling an average of 1.0 LU/ha was applied. The grassland areas were calculated individually for the WFD waterbody and for the extended area (10 km transport bands). The resulting total area LU value were multiplied

Table 7. Typical dry matter and fresh weight nutrient contents of livestock manures and other bulky organic fertilisers (taken from SRuC, 2015).

Manure type	DM (%)	kg/t (solid manures) or kg/m³ (liquids/slurries)		
		Total N	Total P_2O_5	Total K_2O
Cattle FYM	25	6.0	3.2	8.0
Pig FYM	25	7.0	6.0	8.0
Layer manure	35	19	14	9.5
Broiler/Turkey litter	60	30	25	18
Cattle slurry	6	2.6	1.2	3.2
Pig slurry	4	3.6	1.8	2.4
Biosolids, liquid digested	4	2.0	3.0	0.1
Biosolids, digested cake	25	11	18	0.6
Biosolids, thermally dried	95	40	70	2.0
Biosolids, thermally hydrolysed	30	10	20	0.5
Biosolids, lime stabilised	40	8.5	26	0.8
Green compost	60	7.5	3.0	5.5
Green/Food compost	60	11	3.8	8.0
Paper crumble, chemically/ physically treated	40	2.0	0.4	0.2
Paper crumble, biologically treated	30	7.5	3.8	0.4
Distillery pot ale	5	2.5	1.8	1.1
Distillery bioplant effluent/sludge	2.5	1.5	1.3	0.4
Digestate (whole), food based	4	5.0	0.5	2.0
Digestate (whole), pig slurry based	2	3.6	1.8	2.4
Digestate (whole), cattle slurry based	4	2.6	1.2	3.2

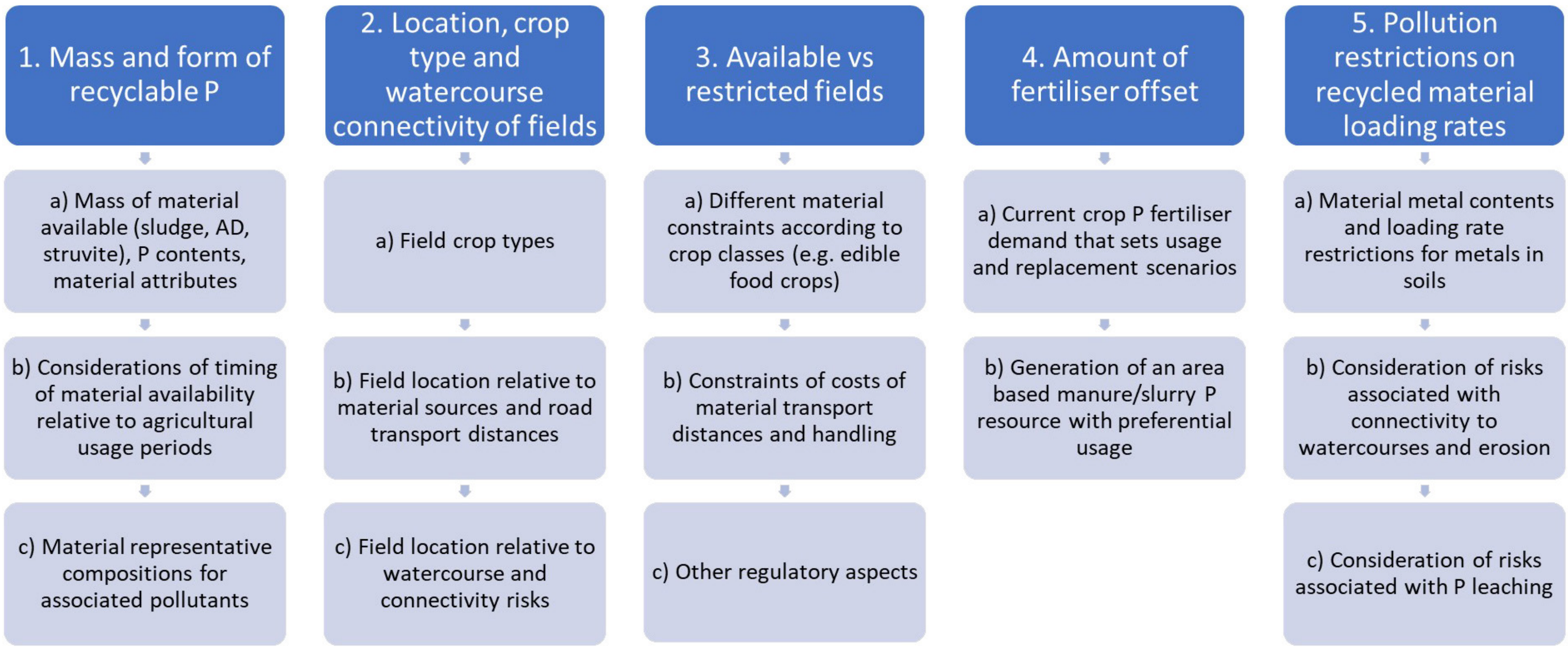


Figure 4. Flowchart of the process of the modelling undertaken in the case study catchments. The steps 1-5 for the basis of the methods sections of this current chapter of this report. Whilst current soil P status is not mentioned here it should be noted that due to specific information on this for fields an effective assumption that the fields were at agronomic optimum was applied, such that the crop P offtakes were used to derive field P requirements on an average annual basis.

by fresh weight values of 0.024 tonnes/day of manure and 0.032 m^3/day of slurry for cows (of 500 kg weight) (FAS, 2018). An estimate of 5 months of indoor housing for cattle in the area was used (150 days). Livestock units capable of manure and/or slurry production were counted on land parcels 2 and 3g (see Tables 9, 10). Outdoor excreta was not calculated as this was assumed to be part of the applications to grazed grassland and so rough grassland (animals mostly out all year) and the periods of outdoor field grazing in land cover codes 2 and 3g were not counted.

The composition of manure (cattle FYM) and cattle slurry was taken from Table 7 (SRuC, 2015) and scaled by the amounts of indoor housed excreta to generate P and N values separately for manure and slurry production within the catchment area.

4.1.2. Considerations of timing of material generation availability

Regarding timing, we wanted to investigate any disconnect between the timing of material generation, availability and its potential use as a fertiliser replacement. Table 8 shows the annual distribution of monthly usage of fertiliser nutrients in agriculture (arable and grassland) within Scotland (Taken from British Survey of Fertiliser Practice, 2019), with the majority being used in cultivation and main growth period of Mar-May and later summer (likely associated with multiple cuts of grass for conserving as winter feed). However, despite this knowledge on use timing it was not possible to gain data on timing of recycled material generation and availability. The generation of both sewage and AD derived recycled P was deemed to be consistent throughout the year and that storage at farms and fields (allowed for up to one year) were sufficient to buffer the relatively narrow annual agricultural usage windows.

4.2. Spatial layout of the case study areas

To characterise the fundamental attributes of the WFD waterbody catchment and the surrounding areas the channel network and field structure were represented using the Ordnance Survey Mastermap layer (https://www.ordnancesurvey.co.uk/documents/os-mastermap-water-network-technical-specification.pdf) and analysis carried out in Arc GIS. Subsequently for specific purposes three groups of analyses were carried out, namely:

(i) accurate field management including cropping for understanding crop P usage and constraints of P usage was derived using Scottish Government's agricultural census data under the Integrated Administration and Control System annual crop reporting (IACS; part of the Common Agricultural Policy (CAP) reporting: www.gov.scot/Topics/farmingrural/Agriculture/grants/Schemes/18148/11836). The IACS claims are made based on land parcels that approximately related to the field layout in divided fields areas but also included larger areas of semi-natural land parcels under grazing land cover categories.

(ii) transport distances between the material source locations and the fields by road;

(iii) risk factors associated with physical attributes of the fields (slope, erosion, proximity to watercourses).

4.2.1. Field crop types

The IACS database for 2015 was queried for field crop codes. The fields were given a project specific identifier to break the personal link between their characteristics and the farm returns. Additionally, no scale of mapping is presented for the raw data or categorised data that can be attributed back to individual farms and locations. Field crop codes were taken as the dominant code for the field area where multiple codes were returned. This yielded 157 codes for differing arable crops, 8 grassland codes and 40 additional codes ruled out on the basis they were non-crop land uses (farm woodland, water etc). The processing requirements for these data necessitated that only one representative year was examined.

The categorisation by a reduced set of crop classes followed a system derived in Balana et al (2012) and given here in Table 9(a). This divided crops into 5 levels for P export risks and their P export coefficients associated with typical management cultivation and fertiliser practices (with additional risk according to field slope). In the current application the classes were extended (Table 19(b)) to satisfy requirements of separation of directly edible, vegetable and combinable food crops (section 4.3.1) and to inform specific classes of erosion risk (section 4.5.2).

Table 8. The nutrient use in chemical fertiliser products as a % of the month of application mass for Scotland in 2019 (British Survey of Fertiliser Practice, 2019; Table SC3.0).

row %	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Nitrogen	1	1	0	0	0	3	15	46	22	8	4	1
Phosphate	7	7	1	0	0	2	17	43	11	4	3	5
Potash	6	7	1	0	0	2	15	46	11	6	3	3
Sulphur	1	1	0	0	0	9	23	45	14	5	2	1
Total	3	3	0	0	0	3	16	45	17	7	4	2

Table 9. Crop risk classes derived used to simplify the complexity of the >200 crop classes in the IACS returns.

(a) original classification from Belana et al (2012)					(b) extended classification
Slope risk class	Land use/crop management	1	2	3	
Mean slope		<4°	4-13°	>13°	
Slope descriptor		Low	Med	High	
Crop risk classes		P export coefficients (kg/ha/year)			
1 (very low)	Rough grazing	0.01	0.02	0.03	Not further modified
2 (low)	Grass over 5 years, fallow, set aside	0.06	0.1	0.14	Not further modified
3 (moderate)	Spring barley, grass under 5 years, spring oats, spring wheat, fodder grass, grass for mowing, fruit	0.3	0.5	0.7	3, combinable cereal crops; 3a, edible food not generally cooked; 3g, grass (<5 years); 3osr, oil seed rape (spring)
4 (high)	Winter barley, winter wheat, peas/beans, winter oats	0.7	1.1	1.5	4, combinable cereal crops; 4b, edible food generally cooked first; 4osr, oil seed rape (winter)
5 (very high)	Turnips/swedes, bulbs/flowers, fodder roots, ware potatoes, seed potatoes, other vegetables	1.3	2.2	3.1	5a, edible food not generally cooked; 5b, edible food generally cooked first

4.2.2. Field distances to recycled material sources and to watercourses

The distances between fields and the locations of Dunnswood WWTW and Cumbernauld AD plant were derived in Arc GIS using the road transport function. For the analysis restricted to the Bonny Water WFD waterbody this was done as a continuous distance ascribed to each field. For the wider area analysis up to 60 km this was done by categories of transport distance (0-10, 10-20, 20-30, 30-40, 40-50 and 50-60 km). Export connectivity to waters was varied according to a riparian vs non-riparian field index derived by applying a 20 m mask to either side of the channel network and categorising fields that intercepted this mask as riparian.

4.3. Available vs restricted fields for offsetting of conventional fertiliser

4.3.1. Constraints of directly edible food crops

For this the guidance of the Safe Sludge Matrix was used (Figure 3) and related to the crop codes derived (Table 19b). In the safe sludge matrix enhanced treated sludges are subject to 10 months harvest interval across all food crops. However, conventionally treated sludges have extended harvest intervals and separation between directly edible and cooked food crops, respectively 30 months and 12 months. These codes related to our crop codes of 3a, 4a, 5a for directly eaten crops (salad and fruit) and 4b, 5b for crops cooked before being eaten. Otherwise, combinable and animal feed crops have no harvest window for conventional and enhanced treatment sludges.

The resulting rules for use of materials on the crop classes was followed (in order of increasing controls on material use):

- Rough grassland (class 1) and on farm woodland – no P inputs
- Intensive grazing grassland (class 2) – AD allowed (3-week harvest interval); no conventional sludge.
- Fodder grass (class 3g) - AD allowed (3-week harvest interval); conventional sludge allowed (assuming no grazing that year).
- Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5) – AD and conventional sludge allowed.
- Human food crops eaten after cooking (class 4b, 5b) - AD allowed (10-month harvest interval); no conventional sludge (harvest intervals considered too long).
- Human food crops, eaten without cooking (class 3a, 5a) – AD allowed (10-month harvest interval); no conventional sludge (harvest intervals considered too long).

4.3.2. Constraints of material transport distances

These calculations used a transport cost of £3.50 per 10 miles per tonne of material for liquid materials (liquid or whole digestate) and £2.50 per 10 miles per tonne of solid digestate (and assumed same for biosolids). An additional spreading cost of £4 per tonne was applied for spreading or injecting of wet digestate or slurries and £2 per tonne for broadcasting of biosolids or manures. The values were taken from WRAP (2016).

4.3.3. Constraints of other regulations

Manures and slurries: Can be applied to agricultural land under the PEPFAA guidance (Prevention of Environmental Pollution from Agricultural Activities) up to the upper N threshold of 250 kgN/ha/year or 50 tonnes/ha.

Biosolids from sewage sludge are not regulated under the Waste Management Licensing Regulations but are regulated instead by the Sludge Use in Agriculture Regulations (Amended, 1990) that place limits on the build-up of heavy metals in soils and prevent application to land where fruit or vegetable crops are growing. Under these regulations, sludge should be used according for the nutrient needs of the crop. Producers are required to maintain records of applications and match those to soil testing required every twenty years. The Safe Sludge Matrix guides the use of sludges on certain crop types and management of land after spreading, for example restricting grazing following application.

Anaerobic digestate: Digestates meeting the requirements set out in the BSI PAS110 scheme are not classed as wastes and can be applied to agricultural land without waste management controls. Application to land should follow guidance set out in the SEPA position statement on digestate use.

Technical guidance (SRuC Technical note TN668 and Defra RB209) for application of fertilisers to crops encourages raising the soil P levels for soils that are below moderate soil P status (SRuC soil analysis method) or index 2 (ADAS method; RB209). The amounts of P fertiliser that can be applied depend on the P status and sorption capacity of the soil alongside the crop P requirements. If the soil P status is above moderate (SRuC soil analysis method) or index 2 (ADAS method; RB209) then the maximum P application rate, calculated based on total, not available, P in the material applied, should be equal to the uptake requirements of the crop. If growing P-demanding crops (potatoes, field vegetables and maize) these soils P levels can be increased to a maximum soil P status of very high, or index 4 under the SRuC and ADAS scales, respectively. When using digestate or biosolid this assumes that 50% of the added P will be available to the crop in the following year. In addition, N application rates should not exceed the limit of 250kgN/ha/year in designated Nitrate Vulnerable Zones and it would be good practice to adhere to this generally in all regions. Specific to anaerobic digestate guidance is given in SEPA 2017b.

4.4. Amount of fertiliser suitable for offsetting

4.4.1. Deriving current P fertiliser inputs to field crops

The fundamental national ethos for P fertiliser use, including that from organic materials, is that the fertiliser application rate matches that of crop requirements. Here we only considered inputs to cropland that matched the

Table 10. Summary of data used to derive crop chemical fertiliser P usage according to crop classes used in the present study. All units as kgP_2O_5/ha/year, except where stated.

Crop class (see Table 9)	Crop examples	P offtakes in Scottish cropping[1]	Average Scottish chemical fertiliser field rates 2019[2]	Final values used	
				kgP_2O_5/ha/ year	kgP/ha/ year
1	Rough grazing			0	0.0
2	Intensive grazing, grass over 5 years	2-3	19	19	8.3
3	Spring cereals	43-69	49	49	21.4
3a	Fruit (currants, berries)			42	18.3
3g (i)	2 cuts of silage	30-41	30	27	11.8
3g (ii)	Hay	36-59	24	27	11.8
3osr	Oil seed rape (spring)	23-38		30	13.1
4	Winter cereals, flowers	59-84	59-61	60	26.2
4b	Peas and beans			42	18.3
4osr	Oil seed rape (winter)	45-76	54	54	23.6
5	Fodder root crops, maize		42	42	18.3
5a	Lettuce			42	18.3
5b (i)	Potatoes		117	117	51.1
5b (ii)	Brassicas, root veg, other veg			42	18.3
na	Farm woodland, no claim fields			0	0.0

Data sources: [1]SRuC Technical note TN668; [2]British Survey of Fertiliser Practice (2020) for 2019 using the average rates of application to fields where applied. Note 42 kgP_2O_5/ha/year was the value assigned to 'other crops' and this value was used where the specific crop was not stated.

crop offtakes at an annual resolution. Although (as noted above) adjustments to increase soil P status for P-depleted soils can be made and reduction of fertiliser on high P soils must be made, in the absence of specific soil P status data for the model areas we assumed a P application to equal crop offtake serves as the average longer-term indicator of usage.

4.4.2. Considerations of existing offsetting of conventional fertiliser using manure

Manure and slurry P availability was calculated according to the methods in section 4.1.1. This calculation derived that material from overwintered livestock where this would be recycled normally back to fields. Grazing excreta returns to grassland are already accounted for in the grassland P fertiliser rates (Table 20). The rule for utilising manures and slurries was that it was used preferentially on land in the order: potatoes (where occurring), cereal crops, other combinable crops, other vegetable crops. Up to half the P requirements for a crop could be used before moving into the next category.

In terms of the spatial rules for utilisation the assumption was that mixed farms effectively used manures and slurries locally. However, scales of transport were not well-understood and the structures of field cropping within aggregated sets of fields making a farm was not examined as were not in-field rotations across years. Because of this a simplified spatial coupling of manure usage was examined within each 10 km transport bands. For example, manure was applied to crops using the above rules in the 10-20 km distance band and this same rule was followed within the surrounding distance bands but not exchanging between bands.

4.5. Pollution restrictions on recycled material usage

4.5.1. Material metal compositions leading to maximum application rates of materials to soils

It is relevant to know the limits on the material applications to agricultural fields placed by the maximum permissible metal loadings to soils. In turn this brings a limit in terms of the maximum P application rates before the allowed average annual metal thresholds are transgressed. This was derived (Table 21) from the representative metal concentrations for biosolids, AD and struvite (Tables 10, 15 and 11, respectively) and related to the application thresholds of the Sludge Regulations in Agriculture 1989 (amended) (Table 16). This was subsequently scaled to field applications on a fresh

Table 11. Calculation of representative maximum loadings of the three main potential fertiliser P replacement materials considered, based on maximum average annual loadings of metals. The metals and loads giving the strictest application rate limits are indicated in bold, then used to set equivalent maximum annual P application rates.

	Max permissible metal addition rate [1]	Material metal contents			Max annual application rates		
		Content in dewatered biosolid [2]	Content in AD [3]	Content in struvite [4]	Biosolid	AD	Struvite
	(kg/ha/year)	(mg/kg dry matter)			(tonnes fresh weight/ha)		
Zinc	15	873	192	201	**69**	**1953**	136
Copper	7.5	433	96	80	**69**	**1953**	170
Nickel	3	41	24	14	296	3125	390
Cadmium	0.15	0.7	0.72	0.85	857	5208	321
Lead	15	50	96	22	1200	3906	1240
Mercury	0.1	0.5	0.48	2.2	800	5208	**83**
Chromium	15	137	48	21	440	7813	1299
Molybdenum	0.2		nd				
Selenium	0.15		nd				
Arsenic	0.7		nd				
Fluoride	20		nd				
Representative dry matter (%)		25%	4%	55%			
Max P addition rate (kgP/ha)					474	860	10495

[1]Maximum addition rates are taken from Sludge Regulations in Agriculture 1989 (amended), here given as Table 6; [2]Representative content in dewatered sludge taken from Table 5, this report (using middle of the stated range); [3]Considers the N content of AD of 5.5 kgN/tonne fresh weight (Table 15, where 'nd' denotes not determined there); [4]Representative contents in struvite derived from Moeller et al (2018; given here Table 5) taking the midpoint of the given range.

weight basis. The strictest limits were associated with potential for zinc and copper in biosolids and AD and for mercury in struvite. It should be noted that associated maximum P loadings of 474, 860 and 10495 kgP/ha/year for biosolids, AD and struvite are excessive compared to field application rates intended to offset crop uptake and possible only to be exceeded in cases of dumping of materials onto land exceeding the rules for crop requirements and soil P status. As a result, we applied no restrictions for maximum metal loadings to soils as the restrictions set in scenarios for P crop utilisation resulted in stricter rules for maximum application rates.

4.5.2. Phosphorus source risks

The original project envisaged modelling the water pollution risk in response to the application of P-rich materials to land for agricultural improvement and specifically to develop simple procedures to quantify the change in P exports from fields and the implications for the freshwater environment when recycled P materials replaced chemical P fertiliser. However, it became apparent that this was not possible for several reasons:

- The knowledge required to understand the relative risks of P movement by erosion and leaching pathways between chemical fertiliser and recycled materials is too underdeveloped at present.
- The agricultural land bank of the target WFD waterbody area was overwhelmed with the amount of P mass in materials produced with the catchment such that the modelling had a much greater sphere of land bank usage that broke the link between the application and a single watercourse catchment.
- In addition, the waterbody for which the SEPA water characterisation data existed comprised a WFD 'segment' (i.e. an incomplete catchment for a reach section that does not include the upper catchment) making changes in the waterbody the product of changes in the P exports in the WFD waterbody and those of an uncharacterised upper catchment.

As a result we undertook a risk based assessment where various levels of information were compiled in order to categorise risks factors associated with the fields of P replacement in order that prioritisation in terms of replacement of chemical fertiliser, could be given to lower, then increasing risk with each transport band. Firstly, the risk factors then the scenarios of this for the case study area are described below.

Risk level 1: Field proximity to watercourse

This utilised the separation of riparian and non-riparian fields, already described (section 4.2.2).

Risk level 2: Inherent erosion risk with crop management factors superimposed

This approach combined assessment of the inherent erosion associated with the soil (texture) and landform (slope) with superimposed risk adjustment according to land management. For example, crop management with a mediating (e.g. permanent grass cover) or exacerbating (e.g. intense cultivation like arable root crops) shifted the inherent erosion risk categories according to Table 12. This procedure has already been embedded into a RAG P assessment framework on phosphorus application to land (Gagkas et al., 2019). Field slopes were determined as the median slope of the field area.

Risk level 3: Soil P leaching risk

In additional to the erosion P risk (level 2, above) we examined within-region differences in the potential for soil P leaching. This was derived by reference to the soil P sorption capacity (PSC) mapping available online (https://map.environment.gov.scot/Soil_maps/) as derived from the soil parent material maps at soil Association level as indicators of the binding strength of P to soil surfaces. The PSC categories of low, medium and high are used in this scale, respectively denoting a lower to stronger degree of binding of P to soil surfaces and corresponding lowering of the risk of P becoming mobile in soil solutions and runoff waters.

4.6. Scenarios of P utilisation

4.6.1. Area and material scenarios

Scenarios were developed considering application areas and a progressive use of differing materials grouped by usage constraint levels. This resulted in three material scenarios and two main area scenarios (the latter subdivided by road transport bands), summarised in Table 13 and described below. These scenarios considered only the agricultural land bank (assuming IACS claim fields represented the 'active' agricultural area). Non-agricultural land was outside of the remit of this study's examination of chemical fertiliser replacement.

Considering firstly the material scenarios, we considered that P bearing materials would replace chemical fertiliser use according to the crop class suitability rules (section 4.3.1) after manures and slurries were consumed locally on appropriate fields (combinable, fodder crops and conserved grassland). Then a prioritisation was given such that Scenario 1 considered the PAS accredited AD material and usage on appropriate fields. Then in Scenario 2 the conventionally treated WWTW sludge was applied to appropriate fields before AD was used on remaining fields (with more stringent crop criteria). In Scenario 3 a limited area of human food crops (crop classes: 3a, 4b, 5a, 5b) were considered for the hypothetical recoverable struvite

Table 12. Derivation of overall erosion risk assessment combining inherent soil erosion risk with management factors (taken from Gagkas et al., 2019). The overall risk assessment is shown as three colours green (low), orange (medium), red (high) for (a) mineral soils in cultivation, and (b) with organic soils used for rough grazing.

(a) Mineral soil erosion risk	Land cover risk (cultivated)		
	Grassland: acid, improved, rough (low)	Arable: cereals (moderate)	Arable: root vegetables (high)
L1	L1 (Low)	L1 (Low)	L2 (Low)
L2	L1 (Low)	L2 (Low)	L3 (Low)
L3	L2 (Low)	L3 (Low)	M1 (Moderate)
M1	L3 (Low)	M1 (Moderate)	M2 (Moderate)
M2	M1 (Moderate)	M2 (Moderate)	M3 (Moderate)
M3	M2 (Moderate)	M3 (Moderate)	H1 (High)
H1	M3 (Moderate)	H1 (High)	H2 (High)
H2	H1 (High)	H2 (High)	H3 (High)
H3	H2 (High)	H3 (High)	H3 (High)

(b) Peat/peaty soils erosion risk	Land cover risk (semi-natural)
	Montane, heath, broadleaves, undisturbed woodland
Li	Li (Low)
Lii	Lii (Low)
Liii	Liii (Low)
Liv	Liv (Low)
Mi	Mi (Moderate)
Mii	Mii (Moderate)
Miii	Miii (Moderate)
H	H (High)

material and excess struvite was considered as a material readily transportable that could be exported from the area to further Scottish land bank (>60 km away, outside of the study area).

Considering the area scenarios, it was necessary to consider the WFD waterbody area and a wider zone. As the investigation progressed the masses of P bearing materials were found to be large compared to the land bank in the WFD waterbody and hence the considered areas were:

- Area A. The analysis restricted to the WFD water body area of 52 km^2.
- Area A+B. The analysis expanded to a 60 km transport distance from the central production locations (Dunnswood WWTW and Cumbernauld AD plant, essentially co-located, neighbouring on the same industrial estate). The 60 km expanded range was chosen on the basis that AD transport to farms was subsidised to that distance. Once the land bank in the WFD waterbody catchment is utilised the progressive 10 km bands of area B are progressively utilised.

As a baseline Scenario 0 a case was considered that all land bank in the area received only chemical fertiliser once manures and slurries had been locally used. In reality the current recycling scenario in the area lies somewhere between levels 0, 1 and 2. But level 0 provided a raw material and economic comparison.

Table 13. Scenarios of material and areas considered in modelling the case study area (chapter 5).

	Land bank scenarios	
Materials	Area A: WFD waterbody	**Area A+B*x to y*: WFD waterbody and 10 km wide transport distance bands (*x to y;* up to 60 km max) from Dunnswood industrial estate**
Usage of chemical fertiliser and manure	***Scenario 0 (A)***: An artificial scenario of purely chemical fertiliser use and local use of manure/slurries considering the WFD waterbody area.	***Scenarios 0, 1, 2* (A+B):** These are equivalent scenario rules for material replacement in column to the left.
Usage of P in Energro digestate product	***Scenario 1 (A)***: Assumes preferential use of digestate P according to Safe Sludge rules and replaces chemical fertiliser after local use of manure/slurry.	The difference is that firstly the waterbody area is utilised then progressive bands of transport distance are considered (until the materials are utilised) in the order: Area A
Usage of P in conventionally processed sludge from Dunnswood	***Scenario 2 (A)***: Used as a priority on combinable and animal feed crops and grassland for harvesting for chemical fertiliser replacement after local use of manure/slurry. Digestate then utilised on remaining land according to scenario 1.	Area A + $B_{0\text{-}10}$ + $B_{10\text{-}20}$ (any remaining area not in area A up to 20 km distance) Area A + $B_{0\text{-}10}$ + $B_{10\text{-}20}$ + $B_{20\text{-}30}$ Area A + $B_{0\text{-}10}$ + $B_{10\text{-}20}$ + $B_{20\text{-}30}$ + $B_{30\text{-}40}$
Use of a hypothetical struvite product derived at Dunnswood WWTW	***Scenario 3 (A)***: Used only as chemical fertiliser replacement on human food crops in study area, or considered exported.	Area A + $B_{0\text{-}10}$ + $B_{10\text{-}20}$ + $B_{20\text{-}30}$ + $B_{30\text{-}40}$ + $B_{40\text{-}50}$ Area A + $B_{0\text{-}10}$ + $B_{10\text{-}20}$ + $B_{20\text{-}30}$ + $B_{30\text{-}40}$ + $B_{40\text{-}50}$ + $B_{50\text{-}60}$
Usage of P in conventionally processed sludge from Bonnybridge and Denny	***Beyond current scenarios***: Not considered due to removal from the area.	
P usage from current effluent discharge into WFD waterbody	***Beyond current scenarios***: Currently not justified by land bank demands in this area due to excess of P bearing materials, requires further legislation on max effluent P concentrations permissible.	

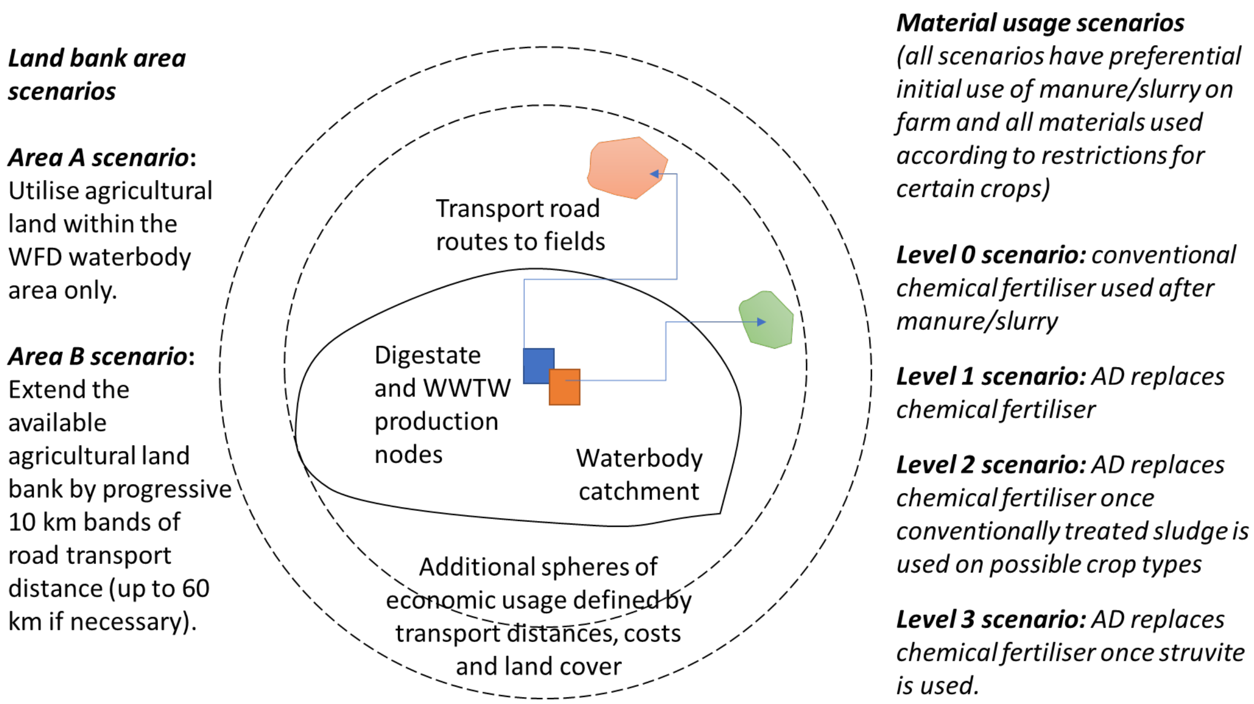


Figure 5. Scenarios of the WFD waterbody (Area A) and extended areas (Area B transport distance bands) and scenarios of the P fertiliser material usage.

4.6.2. Economic analysis of the scenarios

The costs of use of each P-bearing material were calculated for each of the model scenarios, then these were summed as a total scenario fertiliser cost. This cost had three elements:

- The gate price of the material;
- The transport cost of the material – this was done for each of the road transport bands using the mean transport distance (e.g. 10-20 used 15 km distance for all fields);
- The spreading cost associated with the handling (mainly fuel) of the material in the field.

For all the materials these costs are detailed in Table 16. Note that the manure and slurry have assumed gate price and transport costs of zero on the basis that the materials are available for use locally on farm or between adjacent farms. Also, the AD material has no transport costs as this is subsidised and within the gate price. The costs of the different scenarios were compared by summing the necessary cumulative costs of all fertiliser materials and handling costs for the transport bands required to fully utilise the sludge and AD generated at the central production point. The level 0 scenario of chemical fertiliser usage provided the baseline reference cost.

5. Case study results: Bonny Brig catchment

5.1. Supply in the catchment: Mass and form of recyclable P

5.1.1. WWTW derived materials

Four WWTW were identified inside of the catchment of the WFD waterbody and around the catchment boundary. One was considered negligible and three are considered in Table 14. The combined annual inflow loads to these WwTW (2017-19 data) was calculated as 43824 kgP. Of this input 78%, 68% and 64% was apparently removed from the final effluent discharges at Dunnswood (Fig. 6), Bonnybridge and Denny, respectively (as determined by the difference between input and final effluent assumed discharged to watercourses. The fate of the combined 17155 kgP in sludges from Bonnybridge and Denny was to go to Kinneil Kerse sludge treatment centre (Grangemouth). This encompasses transport of 30 km to the plant to mix with other sludges from central Scotland. An aspiration was that here the sludges would be thermally dried to produce a granulated fertiliser product; however, this was not realised.

No information was available on the remaining sludge resource containing 14053 kgP from Dunnswood. The final effluent from Dunnswood discharged 3966 kgP into the WFD waterbody.

Figure 6. Image of Dunnswood WWTW.

Table 14. Sources and fate of local derived P from wastewater processing (data supplied by Scottish Water).

Name of WWTW	Dunnswood, Cumbernauld	Bonnybridge	Denny
Design p.e.	40000	17000	15000
Generated input load p.e.	30493	15188	12115
Lat, long	55.972N, 3.954W	56.009N, 3.869W	56.025N, 3.895W
Discharges into waterbody?	Yes, Bonny water/Red Burn	No, to River Carron	No
Treatment description	Tertiary WwTW with activated sludge treatment and tertiary stage RBC (rotating biological contactor). P removal via ferric sulphate dosing.	Tertiary WwTW with activated sludge treatment. P removal via ferric sulphate dosing.	Tertiary WwTW with activated sludge treatment. P removal via TT-PAX (Taytech).
Load calculations (per day, d)	2019 F&L was 65.37Kg/d. 2017 LIMS was 38.16Kg/d, 2018 LIMS was 59.43Kg/d and 2019 LIMS was 50.50Kg/d. Assumed inlet P of 4.0 mg/l	Average inlet P was 3.28mg/l in 2019. Average inlet flow for 2019 was 13135m3/d. Therefore, average inlet P load was 43.08kg/d.	Average inlet P was 4.62mg/l in 2019. Average inlet flow for 2019 was 5973m3/d. Therefore, average inlet P load was 27.60kg/d.
Annual load of P entering the site (kgP)	18019	15732	10074
Final effluent (FE) discharge P load	2019 F&L was 8.21Kg/d. 2017 LIMS was 6.57Kg/d, 2018 LIMS was 11.14Kg/d and 2019 LIMS was 14.93Kg/d. Assumed inlet conc results in outlet 0.9 mgP/l	Average FE P was 1.45mg/l in 2019. Average FE flow for 2019 was 9,449m3/d. Therefore, average FE discharge P load was 13.70kg/d.	Average FE P was 1.53mg/l in 2019. Average FE flow for 2019 was 6,547m3/d. Therefore, average FE discharge P load was 10.02kg/d.
Annual P export to watercourse (kgP)	3966 to target waterbody	5001 to adjacent waterbody	3650 to adjacent waterbody
Annual mass of P in sludge (inflow load – effluent export) (kgP and % of that entering WwTW)	14053 kgP (78%)	10731 kgP (68%)	6424 kgP (64%)
Composition notes about sludge producing process and fate.	Production and fate not known	Drum thickener on site, poly dosing. Thickened sludge from site transported to Kinneil Kerse STC.	No sludge treatment at Denny, sludge removed from PSTs and transported to Kinneil Kerse STC.

5.1.2. Manures and slurries

The annual production of manures and slurries estimated as produced by overwintered livestock and available for field fertiliser use are given in Table 15 in terms of the P and N resource masses. Farmyard manure and slurry are calculated separately due to differences in the costs of spreading that affects their economic usage. As this is a grassland and mixed farming area (Table 28) the amounts of manure and slurry P resources are large.

5.1.3. Anaerobic digestate

Cumbernauld AD facility (location G67 3EN, adjacent to Dunnswood WwTW) has processed food waste since 2011 and supplies >5500 home equivalents of renewable electricity (https://www.energenbiogas.co.uk/). The capability is for 100000 tonnes of feedstock each year. The feedstocks comprise: animal by-products (cat 3), food industry processing waste, food and drink surplus stock, kitchen food waste and local authority kerbside collection food waste. The producer markets the whole digestate as a liquid bio-fertiliser under the product name Energro (https://www.energro.co.uk/). This is PAS110 accredited, with trace amounts of heavy metals, and is pasteurised against pathogen risks. It is delivered by tanker and expected to be applied by slurry spreader (e.g. band spreader, trailing shoe or injected). Fertiliser value is claimed 8 tonnes equivalent to 200 kg of 22-4-14 NPK and this equates to 0.44 kgP and 44 kgN per tonne FW. The production amounts to approximately 90000 tonnes fresh weight of digestate annually (Energen, pers.comm.). This is used for a fertiliser replacement on ~300 farms locally. The cost at gate price of £2.50/tonne includes a delivery allowance of up to ~60 km, with some farms taking material that are further distances away at additional costs. The gate costs for the material become greater in winter when the plant has storage costs. Due to the 10-month window for harvest of human edible crops it is assumed the digestate is used on a single crop per year but can be applied in every year.

5.1.4. Potential struvite recovery at Dunnswood WWTW

The struvite generation scenario (Fig. 7) shows that 1586 kgP equivalent of struvite is predicted as being possible from the Dunnswood WWTW. The small corresponding reduction in the P content of sludge cake lowers the sludge P resource slightly. But drawdown of effluent P into the struvite has given a predicted reduction in the discharged effluent concentration from 0.9 to 0.8 mgP/L, which would be beneficial for reducing water pollution.

5.1.5. Material availability compared to the national P stocks and flows

At a national level (depicted in Box 1, section 2) there was a modelled 20.4, 9.1, 0.3 and 1.3 ktonnes P annually of chemical fertiliser, animal manure/slurry, AD and sewage sludge, respectively, going to agricultural land in Scotland. This equates to 66%, 29%, 1% and 4% split of those four P fertiliser materials. The available resource of 40000 kgP of AD and 14053 kgP of sludge shows that this area has a large contribution from AD associated with the location of the AD plant at Dunnswood.

5.2. Description of fields in scenario areas

5.2.1. Scenario A fields

Scenario A area concerns the WFD waterbody of 592 fields amounting to 36 km2 (Table 27a). This area itself can be banded by transport distances as approximately half the area being 0-10 km and the remaining half being 10-20 km by road from the Dunnswood WWTW and AD plant site. Approximately a quarter of this area is riparian fields (i.e., bordered by a watercourse). The field areas are dominantly low to medium erosion risk, with an increase in high erosion risk fields in the 10-20 km transport distance band.

Table 15. Manure and slurry resources generated in the considered areas by transport distance bands: (a) WFD waterbody (Area A) and (b) extended area (Area B, excluding areas within the waterbody).

	Total nutrient available per transport distance band (as kgP/year or kgN/year)					
	0-10 km	10-20 km	20-30 km	30-40 km	40-50 km	50-60 km
(a) Area A						
Manure P	6082	5505				
Slurry P	6589	5964				
Manure N	26118	23640				
Slurry N	6964	6304				
(b) Area B (excluding areas already in A)						
Manure P	14930	82706	147897	173745	234770	333597
Slurry P	16175	89599	160222	188244	254334	361397
Manure N	64118	355170	635133	746124	1008187	1432585
Slurry N	17098	94712	169366	198966	268850	382023

Table 16. Material summary of the available P resources and how these are accounted for in modelling. Note that the manure and slurry P resource mass available is representative of that in the WFD waterbody area only (see Table 15 for further details).

	Chemical fertiliser Assumed 20-10-10	Raw manure and slurry	Digestate (Energro product)	Biosolids (Dunnswood)	Final effluent P (Dunnswood)	Struvite creation from Dunnswood	Biosolids (Bonnybridge, Denny)
Actual or assumed material standards	Triple superphosphate	Raw material	Actual: PAS110 (meets enhanced treated sludge rules)	Assumed: Digested cake	Current final effluent raw form	Chemical form $MgNH_4PO_4.6H_2O$	Assumed: Conventionally treated sludges
N content (kgN/tonne FW)		Manure = 6 Cattle slurry = 1.2	5.5	11		57	
P content (kgP/tonne FW)		Manure = 1.40 Cattle slurry = 1.14	0.44	6.89	0.0009 in raw form	127	
Price for material (£/tonne FW)	£540 (at £324 per 600kg bag)	No cost, by-product	£2.50 gate price to farmers, includes transport cost within 60 km	Estimated £21 cost to WWTW for removal, stockpiling and spreading.		Production costs assumed neutral. Gate price £100 indicative.	
Total FW mass available (tonnes/year)	Assumed unlimited		90 000	Fate stated as not known, assumed available to use	Presently 'lost' by discharge as effluent		Assumed 'lost' by transport out of the area
Total P resource available (kgP/year)	Assumed unlimited	**24139** (in WFD waterbody area)	40 000	14 053	3966	**1586** (based on 8.8% conversion of total inflow P)	**17 155**
Gate price £/kgP	**£14.15**	**No cost**	**£5.68**	**£2.00** (assumed)		**£0.81** (assumed)	**Not known**
Constraints of composition according to Safe Sludge Matrix	None above requirement to match crop offtakes	Shouldn't exceed 50 tonnes/ha	Allowable on combinable and animal feed crops, also on grassland and on field vegetables and salads with harvest interval.	Conventionally-treated sludge: Allowable only on combinable and animal feed crops and grassland for harvesting.		None above requirement to match crop offtakes	Allowable only on combinable and animal feed crops and grassland for harvesting.
If apply to N max then what is the max P application rate (kgP/ha/year)		Manure = 58 Cattle slurry = 237	2.5	157		557	
Handling weight of each kgP (tonnes FW)	0.03	Manure = 0.7 Cattle slurry = 0.9	2.3	0.15	1111 in raw form	0.008	
Transport costs per kgP (£/km)	<£0.01	No cost, considered on farm already	£0.49, included in price	£0.04	£173 in raw form	<£0.01	
Spreading costs per kgP (£)	£0.05	Manure = £2.80 Cattle slurry = £4.54	£9.10	£0.30	£4444 in raw form	£0.02	

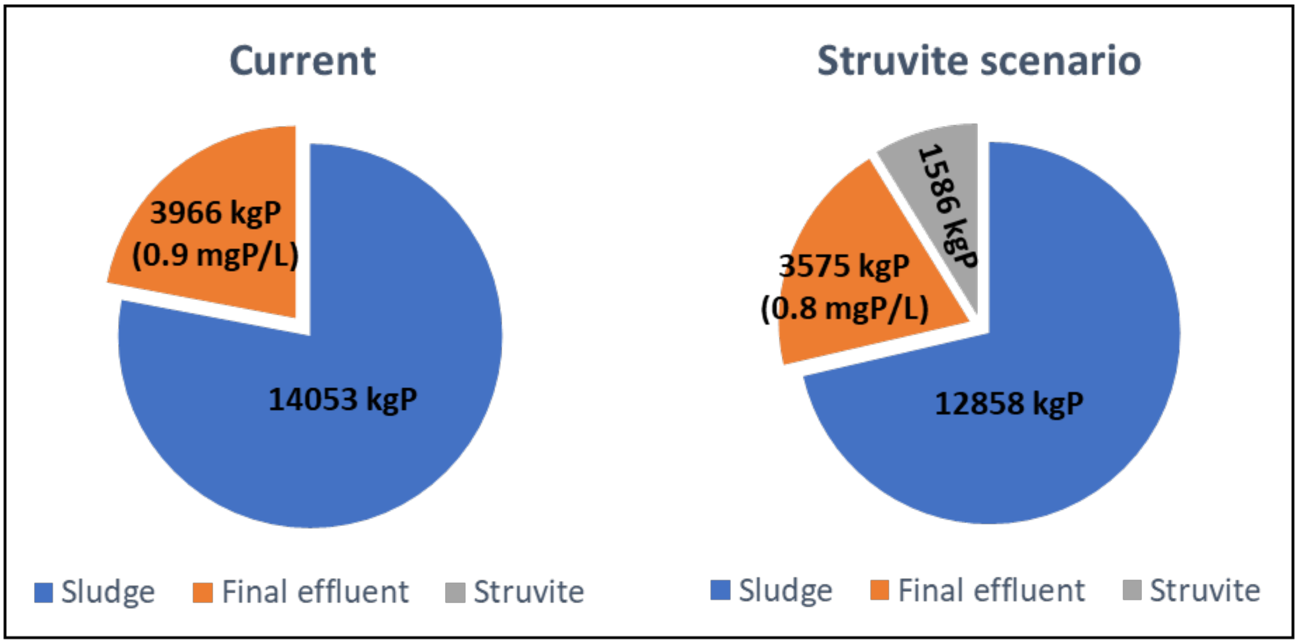


Figure 7. The struvite potential generation scenario at Dunnswood (based on data from the Phosphaq and Ostara processes; Kleeman et al., 2015).

5.2.2. Scenario A+B fields

The scenario A+B examines the land bank of the WFD waterbody then progressively the areas outside of this in 10 km road transport bands. The description of these areas in Table 17b excludes any areas that fall within the WFD waterbody. The analysis has included many fields to examine the area to which AD is delivered with subsidised transport. The distribution of riparian and non-riparian fields in the expanded area is like in the WFD waterbody. In terms of erosion risk the highest risk areas occupy 3-33% of the field areas.

Table 17. Summary statistics of the fields in the case study areas by transport bands and giving both field numbers (n) and combined areas (km^2) as total values for all fields in the areas, then the % distribution between (i) riparian and non-riparian fields and areas, then (ii) erosion risk low (L), medium (M) and high (H) areas.

	Transport distance bands (km)											
	0-10		10-20		20-30		30-40		40-50		50-60	
	n	km^2	n	km^2	n	km^2	n	km^2	n	km^2	n	km^2
(a) Area A												
Total fields	397	19	195	17								
% Riparian fields	18	21	24	29								
% Non-riparian	82	79	76	71								
% Erosion risk L	43	45	48	41								
% Erosion risk M	55	45	38	24								
% Erosion risk H	3	10	13	35								
(b) Area B (excluding in area A)												
Total fields	1986	111	8825	613	15396	1200	18131	2064	24317	2336	33101	2771
% Riparian fields	17	23	22	24	20	25	19	25	17	24	18	22
% Non-riparian	83	77	78	76	80	76	81	75	83	76	82	78
% Erosion risk L	63	66	60	49	54	40	51	28	51	30	42	27
% Erosion risk M	36	32	36	37	41	37	43	39	43	43	53	55
% Erosion risk H	1	3	4	13	5	23	6	33	6	27	5	18

Table 18. Summary statistics of the crop fertiliser demands (ie inputs that would match annual offtakes) by crop categories in the case study areas by transport bands. Crop categories in brackets relate to those in Tables 9 and 10 and are grouped according to the different rules of P material application used in the Safe Sludge Matrix and here in the material scenarios 1, 2 and 3.

	Sum of P demand as area (km^2) then annual total P (kg)											
	0-10		10-20		20-30		30-40		40-50		50-60	
	km^2	Kg	km^2	Kg	km^2	Kg	km^2	Kg	km^2	Kg	km^2	Kg
Area A												
Rough grassland (class 1) and farm woodland	4.4	0	3.6	0								
Intensive grazing grassland (class 2)	9.7	8018	9.9	8210								
Fodder grass (class 3g)	2.4	2870	1.1	1242								
Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	2.7	6036	0.08	167								
Human food crops eaten after cooking (class 4b, 5b)	0.1	1274	0	0								
Human food crops, eaten without cooking (class 3a, 5a)	0	0	0.1	236								
Area B (excluding in area A)												
Rough grassland (class 1) and farm woodland	53.7	0	399	0	703	0	1472	0	1694	0	1816	0
Intensive grazing grassland (class 2)	17.4	14426	121	100094	191	158280	248	206141	307	254579	445	369353
Fodder grass (class 3g)	12.3	14518	43.8	51726	103	121941	97.1	114537	160	188836	218	257256
Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	23.0	53661	60.3	142688	136	324284	128	300719	169	393831	273	632385
Human food crops eaten after cooking (class 4b, 5b)	0.1	1330	0.1	1243	0.9	9979	4.5	52853	6.1	71530	16.9	198239
Human food crops, eaten without cooking (class 3a, 5a)	0.1	129	0.1	129	0.1	131	0.1	227	0.5	930	1.4	2605

5.3. Scenarios of P fertiliser replacement in the fields

The scenarios are described in the sections below. The general land bank available for fertiliser replacement is described in Table 18 and depicted in Fig. 8 in terms of the area distribution amongst the grouped crop classes and in Fig. 9 according to the kgP requirements associated with those land areas.

5.3.1. Fertiliser replacement in the WFD waterbody (Area A scenarios)

Material scenario 0: Chemical fertiliser usage after manures and slurries used on appropriate land

The scenario of chemical fertiliser use is presented in Table 19a. For this annual scenario available manure and slurry provides an excess of 6430 and 7396 kgP, respectively,

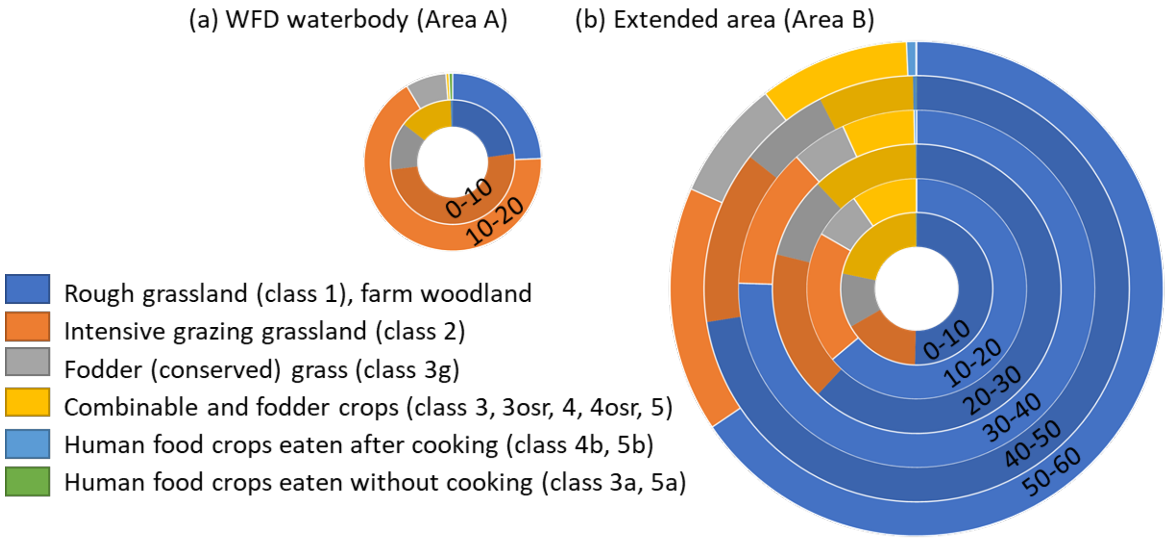


Figure 8. Distributions of field areas in crop class groupings (proportions of colour within a circle) shown by road transport distance bands for (a) the WFD waterbody (Area A) and (b) the extended area (Area B, excluding any area already in Area A). Derived from sum of field areas in km^2.

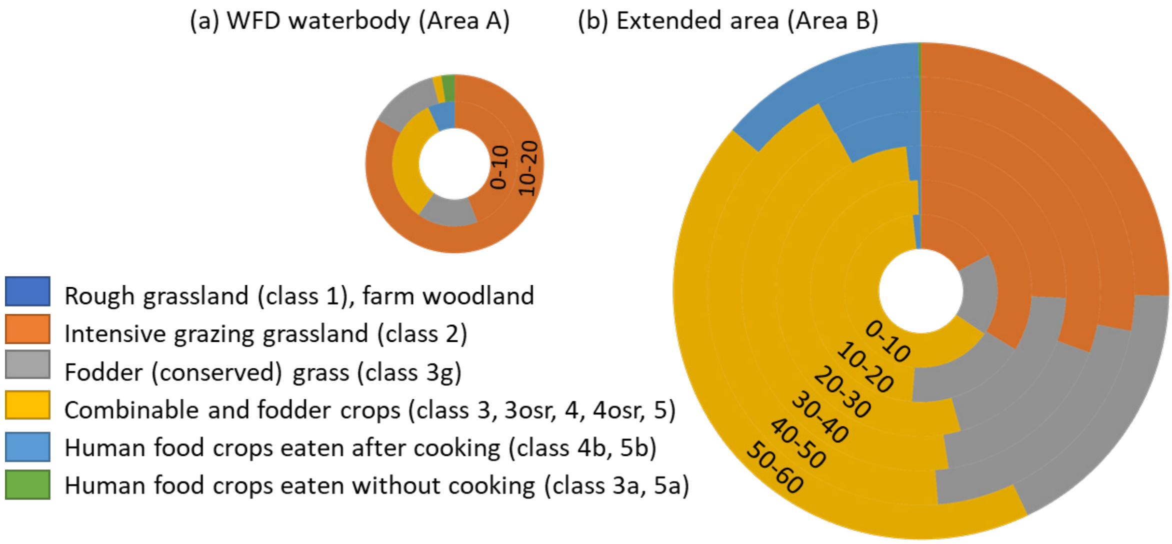


Figure 9. Distributions of crop P requirements between crop class groupings (proportions of colour within a circle) shown by road transport distance bands for (a) the WFD waterbody (Area A) and (b) the extended area (Area B, excluding any area already in Area A). Derived from sum of field P usage requirements in kgP.

once it is utilised on the appropriate available land bank (shaded grey crop classes in Table 19). Then 17738 kgP of chemical fertiliser is used on the remaining fields. The total fertiliser cost of this scenario is £291K (dominated by the chemical fertiliser gate price).

Material scenario 1: AD replaces chemical fertiliser usage after manures and slurries used on appropriate land

The scenario of AD replacing chemical fertiliser use is presented in Table 19b. For this annual scenario available manure and slurry is in the equivalent excess (as in Scenario 0) of 6430 and 7396 kgP, respectively, once it is utilised on the appropriate available land bank (shaded grey crop classes in Table 19). Then 17738 kgP of AD is possible to entirely replace chemical fertiliser on the remaining fields. The total fertiliser cost of this scenario of £300K, only a little greater than for the chemical fertiliser costs in the Scenario 0.

Material scenario 2: Conventional sludge use on combinable and animal fodder crops, then AD replaces chemical fertiliser usage after manures and slurries used on appropriate land

This scenario was not possible. The excess of manure and slurry means that there is no capacity in the allowable fields for use of conventionally treated sludge.

Material scenario 3: Utilisation of a hypothetical struvite product on human food crop land where other recycled inputs are restricted

The scenario of struvite replacing chemical fertiliser in human food crops, then AD replacing remaining chemical fertiliser use is presented in Table 19c. For this annual scenario available manure and slurry is in the equivalent excess (as in Scenario 0) of 6430 and 7396 kgP, respectively, once it is utilised on the appropriate available land bank (shaded grey crop classes in Table 19). Then 1510 kgP, close to the sum hypothetical availability of 1586 kgP struvite available from enhanced processes at Dunnswood WWTW can be used. This struvite use replaces AD use such that the excess of AD increases. However, since the gate price used in the modelling of struvite is low the total area fertiliser costs is £279K. This presents a cost reduction of £21K annually compared to fully AD utilisation, or a cost saving of £12K annually compared to the reference conventional chemical fertiliser scenario 0.

5.3.2. Fertiliser replacement in the extended area (Area A+B scenarios)

In this scenario the area A representing the WFD waterbody is supplemented by a further area up to 20 km road-based material transport distance (area B). Since the WFD waterbody comprised fields within transport distance bands 0-10 km and 10-20 km it was decided to undertake the combined 0-10 and 10-20 km transport bands of area B into one scenario. Although the land bank statistics for transport bands up to 60 km are presented (Tables 27, 28; Fig. 8, 9) it was subsequently found that the centrally produced P bearing fertiliser replacement materials could be utilised within the extended area B considering up to 20 km distance. Therefore, Table 20 presents the land bank utilisation of the different scenarios of P materials up to 20 km only. In the case of extended area B there was a deficit of manure and slurry relative to land bank fields where these fertilisers could be applied and hence the scenario 2, with utilisation of conventionally treated WWTW sludge, could be applied, where this had not been possible in considering the WFD waterbody only (Table 19).

Material scenario 0: Chemical fertiliser usage after manures and slurries used on appropriate land

The extended area scenario of chemical fertiliser use is presented in Table 20a. For this annual scenario available manure and slurry providing 227550 kgP, could be fully utilised on the appropriate available land bank. Then 180447 kgP of chemical fertiliser is used on the remaining fields. The total fertiliser cost of this scenario is £3427K.

Material scenario 1: AD replaces chemical fertiliser usage after manures and slurries used on appropriate land

The scenario of AD replacing chemical fertiliser use is presented in Table 20b. For this annual scenario manure and slurry was again fully utilised (as in scenario 0). Then it was possible to utilise all of the 40000 kgP of AD within the extended area provided by A and B within the maximum of the 10 km road transport band. This left a remaining usage of chemical fertiliser of 18486 kgP and 121961 kgP within the 0-10 km and 10-20 km transport band distances, respectively. The total fertiliser cost of this scenario of £3448K, only a little greater than for the chemical fertiliser costs in the Scenario 0.

Material scenario 2: Conventional sludge use on combinable and animal fodder crops, then AD replaces chemical fertiliser usage after manures and slurries used on appropriate land

The scenario of conventional sludge, then AD, replacing chemical fertiliser use is presented in Table 20c. In this scenario the full amount of on-farm produced manure and slurry was utilised and then the 40000 kgP of AD was also fully utilised. The larger crop area of combinable and fodder crops in the extended area then allowed the conventionally treated sludge to be utilised on this land bank (within the 10 km transport threshold). This had a total scenario fertiliser cost of £3283K and was the cheapest of the area A+B scenarios.

Table 19 (a to c) (overpage). Calculations towards P fertiliser usage modelling in the WFD waterbody area (Area A) considering (a) the level 0 (chemical fertiliser usage), (b) the level 1 scenario of AD usage replacing chemical fertiliser, and (c) the Level 3 scenario of struvite usage on human food crops. In all cases manure and slurry is preferentially utilised first. Grey shaded boxes for crop classes denote where the P bearing material can be used. Level 2 scenarios were not possible as no land bank for wastewater sludge was available.

(a)			Manure	Slurry	Chemical fertiliser	AD	sludge	Total P required
	Input	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**14053**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	6082	6589				
	Usage	Rough grassland (class 1), farm woods			0			0
	Usage	Intensive grazing grassland (class 2)			8018			8018
	Usage	Fodder grass (class 3g)	1435	1435	0			2870
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	3018	3018	0			6036
	Usage	Human food crops eaten after cooking (class 4b, 5b)			1274			1274
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			0			0
	Excess	Manure, slurry excess	1629	2136				
	Cost	Cost of usage: gate price (£)	0	0	131482			
	Cost	Cost of usage: transport price (£)	0	0	465			
	Cost	Cost of usage: spreading price (£)	12468	20217	465			
	Transfer	**Input transferred to next distance (kgP/year)**						
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**5505**	**5964**				
	Usage	Rough grassland (class 1), farm woods			0			0
	Usage	Intensive grazing grassland (class 2)			8210			8210
	Usage	Fodder grass (class 3g)	621	621	0			1242
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	84	84	0			167
	Usage	Human food crops eaten after cooking (class 4b, 5b)			0			0
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			236			236
	Excess	Manure, slurry excess	4801	5260				
	Cost	Cost of usage: gate price (£)	0	0	119511			
	Cost	Cost of usage: transport price (£)	0	0	1267			
	Cost	Cost of usage: spreading price (£)	1973	3198	422			
Summary	**Input**	**Material excess overall (kgP/year)**	**6430**	**7396**				
	Cost	**Sum of costs per fertiliser material (£)**	**14441**	**23415**	**253611**			
	Cost	**Total cost (£)**						**£291,467**

(b)			Manure	Slurry	Chemical fertiliser	AD	sludge	Total P required
	Input	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**14053**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**6082**	**6589**				
	Usage	Rough grassland (class 1), farm woods			0	0		0
	Usage	Intensive grazing grassland (class 2)			0	8018		8018
	Usage	Fodder grass (class 3g)	1435	1435	0			2870
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	3018	3018	0			6036
	Usage	Human food crops eaten after cooking (class 4b, 5b)			0	1274		1274
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			0	0		0
	Excess	Manure, slurry excess	1629	2136				
	Cost	Cost of usage: gate price (£)	0	0	0	52779		
	Cost	Cost of usage: transport price (£)	0	0	0	0		
	Cost	Cost of usage: spreading price (£)	12468	20217	0	84557		
	Transfer	**Input transferred to next distance (kgP/year)**				**30708**		
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**5505**	**5964**				
	Usage	Rough grassland (class 1), farm woods						0
	Usage	Intensive grazing grassland (class 2)			0	8210		8210
	Usage	Fodder grass (class 3g)	621	621	0			1242
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	84	84	0			167
	Usage	Human food crops eaten after cooking (class 4b, 5b)			0	0		0
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			0	236		236
	Excess	Manure, slurry excess	4801	5260				
	Cost	Cost of usage: gate price (£)	0	0	0	47973		
	Cost	Cost of usage: transport price (£)	0	0	0	0		
	Cost	Cost of usage: spreading price (£)	1973	3198	0	76859		
Summary	**Input**	**Material excess overall (kgP/year)**	**6430**	**7396**		**22262**		
	Cost	**Sum of costs per fertiliser material (£)**	**14441**	**23415**	**0**	**262168**		
	Cost	**Total cost (£)**						**£300,024**

			Manure	Slurry	Chemical fertiliser	AD	struvite	Total P required
(c)	**Input**	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**1586**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**6082**	**6589**				
	Usage	Rough grassland (class 1) and farm woodland			0	0		0
	Usage	Intensive grazing grassland (class 2)			0	8018		8018
	Usage	Fodder grass (class 3g)	1435	1435	0	0		2870
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	3018	3018	0	0		6036
	Usage	Human food crops eaten after cooking (class 4b, 5b)			0	0	1274	1274
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			0	0	0	0
	Excess	Manure, slurry excess	1629	2136				
	Cost	Cost of usage: gate price (£)	0	0	0	45542	1031.9	
	Cost	Cost of usage: transport price (£)	0	0	0	0	64	
	Cost	Cost of usage: spreading price (£)	12468	20217	0	72964	64	
	Transfer	**Input transferred to next distance (kgP/year)**				**31982**	**312**	
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**5505**	**5964**				
	Usage	Rough grassland (class 1) and farm woodland						0
	Usage	Intensive grazing grassland (class 2)			0	8210		8210
	Usage	Fodder grass (class 3g)	621	621	0			1242
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	84	84	0			167
	Usage	Human food crops eaten after cooking (class 4b, 5b)			0	0	0	0
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			0		236	236
	Excess	Manure, slurry excess	4801	5260				
	Cost	Cost of usage: gate price (£)	0	0	0	46633	191.16	
	Cost	Cost of usage: transport price (£)	0	0	0	0	12	
	Cost	Cost of usage: spreading price (£)	1973	3198	0	74711	12	
Summary	**Input**	**Material excess overall (kgP/year)**	**6430**	**7396**		**23772**	**76**	
	Cost	**Sum of costs per fertiliser material (£)**	**14441**	**23415**	**0**	**239850**	**1374**	
	Cost	**Total cost (£)**						**£279,080**

Table 20 (a to d) (overpage). Calculations towards P fertiliser usage modelling in the WFD waterbody and extended area up to 20 km road transport distance (sum of areas $A+B_{0-20\ km}$) considering (a) the level 0 scenario (chemical fertiliser usage reference), (b) the level 1 scenario of AD usage replacing chemical fertiliser, (c) the level 2 scenario of conventional sludge utilisation after use of on-farm manure and slurry, but before use of AD and (d) the Level 3 scenario of struvite usage on human food crops. In all cases manure and slurry is preferentially utilised first. Grey shaded boxes for crop classes denote where the P bearing material can be used.

(a)			Manure	Slurry	Chemical fertiliser	AD	sludge	Total P required
	Input	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**14053**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**21012**	**22764**				
	Usage	Rough grassland (class 1) and farm woodland			0			0
	Usage	Intensive grazing grassland (class 2)			22444			22444
	Usage	Fodder grass (class 3g)			17388			17388
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	21012	22764	15921			59697
	Usage	Human food crops eaten after cooking (class 4b, 5b)			2604			2604
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			129			129
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	827577			
	Cost	Cost of usage: transport price (£)	0	0	2924			
	Cost	Cost of usage: spreading price (£)	58834	103349	2924			
	Transfer	**Input transferred to next distance (kgP/year)**	**0**	**0**				
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**88211**	**95563**				
	Usage	Rough grassland (class 1) and farm woodland			0			0
	Usage	Intensive grazing grassland (class 2)			108304			108304
	Usage	Fodder grass (class 3g)	16784	24136	12049			52968
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	71428	71428	0			142855
	Usage	Human food crops eaten after cooking (class 4b, 5b)			1243			1243
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			365			365
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	1725748			
	Cost	Cost of usage: transport price (£)	0	0	18294			
	Cost	Cost of usage: spreading price (£)	246991	433856	6098			
Summary	**Input**	**Material excess overall (kgP/year)**	**0**	**0**				
	Cost	**Sum of costs per fertiliser material (£)**	**305824**	**537205**	**2583566**			
	Cost	**Total cost (£)**					**£3,426,595**	

			Manure	Slurry	Chemical fertiliser	AD	sludge	Total P required
(b)	**Input**	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**14053**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**21012**	**22764**				
	Usage	Rough grassland (class 1) and farm woodland			0	0		0
	Usage	Intensive grazing grassland (class 2)			0	22444		22444
	Usage	Fodder grass (class 3g)			0	17388		17388
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	21012	22764	15753	168		59697
	Usage	Human food crops eaten after cooking (class 4b, 5b)			2604			2604
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			129			129
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	261577	227200		
	Cost	Cost of usage: transport price (£)	0	0	924	0		
	Cost	Cost of usage: spreading price (£)	58834	103349	924	364000		
	Transfer	**Input transferred to next distance (kgP/year)**	**0**	**0**		**0**		
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**88211**	**95563**				
	Usage	Rough grassland (class 1) and farm woodland						0
	Usage	Intensive grazing grassland (class 2)			108304			108304
	Usage	Fodder grass (class 3g)	16784	24136	12049			52968
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	71428	71428	0			142855
	Usage	Human food crops eaten after cooking (class 4b, 5b)			1243			1243
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			365			365
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	1725748	0		
	Cost	Cost of usage: transport price (£)	0	0	18294	0		
	Cost	Cost of usage: spreading price (£)	246991	433856	6098	0		
Summary	**Input**	**Material excess overall (kgP/year)**	**0**	**0**		**0**		
	Cost	**Sum of costs per fertiliser material (£)**	**305824**	**537205**	**2013566**	**591200**		
	Cost	**Total cost (£)**					**£3,447,795**	

(c)			Manure	Slurry	Chemical fertiliser	AD	sludge	Total P required
	Input	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**14053**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**21012**	**22764**				
	Usage	Rough grassland (class 1) and farm woodland			0	0		0
	Usage	Intensive grazing grassland (class 2)			0	22444		22444
	Usage	Fodder grass (class 3g)			0	17388		17388
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	21012	22764	1700	168	14053	59697
	Usage	Human food crops eaten after cooking (class 4b, 5b)			2604			2604
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			129			129
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	62727	227200	28106	
	Cost	Cost of usage: transport price (£)	0	0	222	0	2811	
	Cost	Cost of usage: spreading price (£)	58834	103349	222	364000	4216	
	Transfer	**Input transferred to next distance (kgP/year)**	**0**	**0**		**0**	**0**	
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**88211**	**95563**				
	Usage	Rough grassland (class 1) and farm woodland						0
	Usage	Intensive grazing grassland (class 2)			108304			108304
	Usage	Fodder grass (class 3g)	16784	24136	12049			52968
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	71428	71428	0			142855
	Usage	Human food crops eaten after cooking (class 4b, 5b)			1243			1243
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			365			365
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	1725748	0	0	
	Cost	Cost of usage: transport price (£)	0	0	18294	0	0	
	Cost	Cost of usage: spreading price (£)	246991	433856	6098	0	0	
Summary	**Input**	**Material excess overall (kgP/year)**	**0**	**0**		**0**	**0**	
	Cost	**Sum of costs per fertiliser material (£)**	**305824**	**537205**	**1813311**	**591200**	**35133**	
	Cost	**Total cost (£)**					**£3,282,672**	

			Manure	Slurry	Chemical fertiliser	AD	struvite	Total P required
(d)	**Input**	**System P inputs: available P bearing material (kgP/year)**			**unlimited**	**40000**	**1586**	
0-10 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**21012**	**22764**				
	Usage	Rough grassland (class 1) and farm woodland			0	0		0
	Usage	Intensive grazing grassland (class 2)			0	22444		22444
	Usage	Fodder grass (class 3g)			0	17388		17388
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	21012	22764	15753	168		59697
	Usage	Human food crops eaten after cooking (class 4b, 5b)			1018		1586	2604
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			129			129
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	239135	227200	1285	
	Cost	Cost of usage: transport price (£)	0	0	845	0	79	
	Cost	Cost of usage: spreading price (£)	58834	103349	845	364000	32	
	Transfer	**Input transferred to next distance (kgP/year)**	**0**	**0**		**0**	**0**	
10-20 km road transport distance	Input	Manure, slurry internally created (kgP/year)	**88211**	**95563**				
	Usage	Rough grassland (class 1) and farm woodland						0
	Usage	Intensive grazing grassland (class 2)			108304			108304
	Usage	Fodder grass (class 3g)	16784	24136	12049			52968
	Usage	Combinable and fodder crops (class 3, 3osr, 4, 4osr, 5)	71428	71428	0			142855
	Usage	Human food crops eaten after cooking (class 4b, 5b)			1243			1243
	Usage	Human food crops, eaten without cooking (class 3a, 5a)			365			365
	Excess	Manure, slurry excess	0	0				
	Cost	Cost of usage: gate price (£)	0	0	1725748	0	0	
	Cost	Cost of usage: transport price (£)	0	0	18294	0	0	
	Cost	Cost of usage: spreading price (£)	246991	433856	6098	0	0	
Summary	**Input**	**Material excess overall (kgP/year)**	**0**	**0**		**0**	**0**	
	Cost	**Sum of costs per fertiliser material (£)**	**305824**	**537205**	**1990965**	**591200**	**1396**	
	Cost	**Total cost (£)**					**£3,426,590**	

Material scenario 3: Utilisation of a hypothetical struvite product on human food crop land where other recycled inputs are restricted

The scenario of struvite replacing chemical fertiliser in human food crops, then AD replacing remaining chemical fertiliser use is presented in Table 20d. The whole of the hypothetical availability of 1586 kgP struvite possible to be derived from enhanced processes at Dunnswood WWTW can be used in this scenario. This struvite use replaces conventional fertiliser use in the extended area (as opposed to replacing use of AD in the more limited WFD waterbody area). The total fertiliser cost of this scenario of £3427K, identical to that for the chemical fertiliser costs in the Scenario 0.

Summary of the offsetting of chemical fertiliser compared to costs overall

The smaller area of the WFD waterbody (area A) has ~60% of the fertiliser P requirements predicted as utilising conventional fertiliser and 40% utilising on farm manures and slurry in the reference (level 0) scenario (Fig. 10a). In the recycled material scenarios 40% of the fertiliser P requirements remain satisfied with manures and the 60% remaining able to switch to use of the AD product, since being accredited this has wide usage ability across crop types. Sludges cannot be used in the WFD waterbody area since they compete with on farm organic fertilisers and there is a net excess of manure, slurry and around half of the AD needed use in the surrounding area. In terms of cost comparisons (Fig. 10c) the reference scenario shows that 86% of the waterbody area fertiliser cost is attributed to the conventional fertiliser dominated by the high gate price (£14/kgP). In the recycled material scenarios this cost transfers to a slightly greater cost for AD that arises from the moderate gate price (inclusive of subsidised transport of £6/kgP) and the high handling costs (£9/kgP).

The scenario using the combined area of the waterbody plus the extended area to the common maximum road transport distance of 20 km has a greatly expanded field area demanding considerably more fertiliser P input. The land use allows all of the manures and slurry in this expanded area to be utilised and this on farm fertiliser usage is the majority potential P fertiliser input (56%) compared to chemical P fertiliser (44%) in the reference (level 0) scenario. Then in the recycled material scenarios progressively all the AD (level 1 scenario), or all the AD plus either the conventionally treated sludge (level 2) or hypothetical struvite (level 3 scenario) are capable of utilisation considering the cropping of the field and the

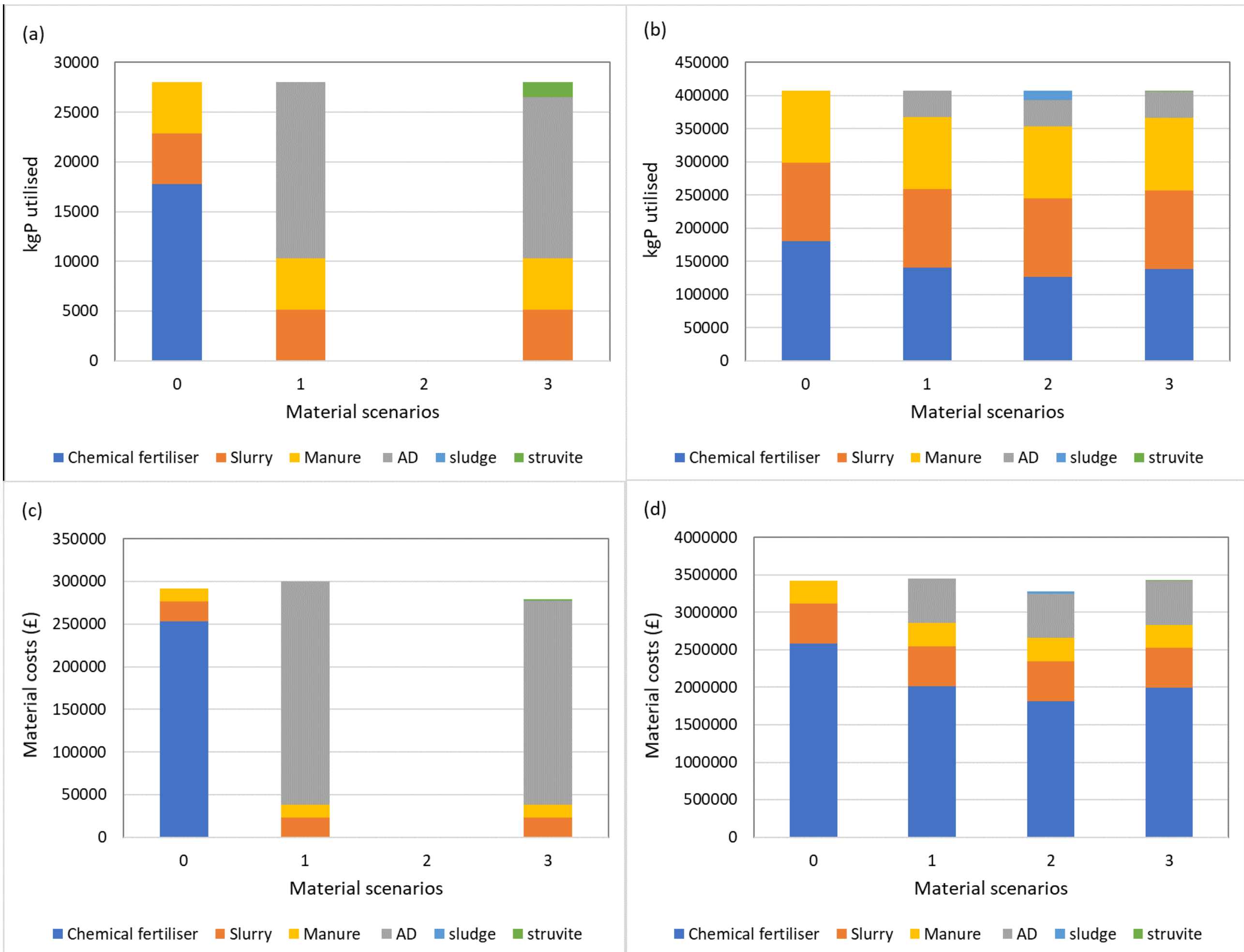


Figure 10. Bar graphs of the distribution of fertiliser materials to their totals under the different material scenarios (levels 0, 1, 2, 3) in terms of kgP (panels a and b) and costs (panels c and d). The panels (a) and (c) show the WFD waterbody Area A and panels (b) and (d) show the extended Area A+B provided by the waterbody plus other fields up to the 20 km road transport distance.

application rules. These recycled material scenarios give up to 13% of the conventional fertiliser. However, this utilisation of the recycled materials was possible within the 0-10 km road transport distance of the area B. If only the WFD waterbody (Area A) and the 0-10 km transport band of Area B was considered the percentage of recycled materials to overall P fertiliser would have been much greater. But since the WFD waterbody had land up to 20 km from the Dunnswood site of central processing it made sense to combine areas A+B up to the 20 km distance band. The costs across the Area A+B scenarios (Fig. 10d) remain dominated by conventional fertiliser gate price costs. However, the costs of AD usage become proportionate to the amount that AD replaces chemical fertiliser as the prices are so similar.

5.4. P replacement and water pollution potential risks

5.4.1. Risk categorisation of the land bank

The risk categorisation comprised three elements, namely: the field proximity to any watercourses (designation of riparian vs non-riparian fields); erosion risk from combination of inherent (soil and landform) and crop management factors; soil P leaching risk.

The variation in the soil phosphorus sorption capacity (PSC) associated with the soil parent materials was used to evaluate the soil P leaching risks. The maps of the soil P risks show that the farmed area is dominated by soil Associations that are in the high PSC class. Some areas of the Clyde valley bottom that are in medium PSC are mostly developed land around roads and towns. A minor area of low PSC is on a hill where upland land use prevails. Hence, this is an area of Scotland where the soils tend towards higher capacities for P sorption without strong regional variation in ability to sorb P that would give variability in P leaching risks spatially (at the available mapping scale derived from soil Associations).

Watercourse proximity and erosion risk factors are depicted in Fig. 11. Considering the WFD waterbody area (Fig. 11a) 81% and 19% of fertiliser P usage was within non-riparian fields and riparian fields, respectively. The former has the lower direct risk of connectivity of P from field runoff entering watercourses. The field having high risks were a small component of the overall P utilisation area, with 8% of P requirements for high risk, non-riparian and 1% for high risk, riparian fields. In Fig. 10b the 0-10 km transport band distance of extended area B was considered for risks on the basis that the recycled materials could be fully utilised in this transport distance without going further from the production site at Dunnswood. Within this area the P usage was potentially to 77% of non-riparian fields and 23% of riparian fields. However, the erosion risks were shifted towards lower risk categories; in this area effectively none of the potential

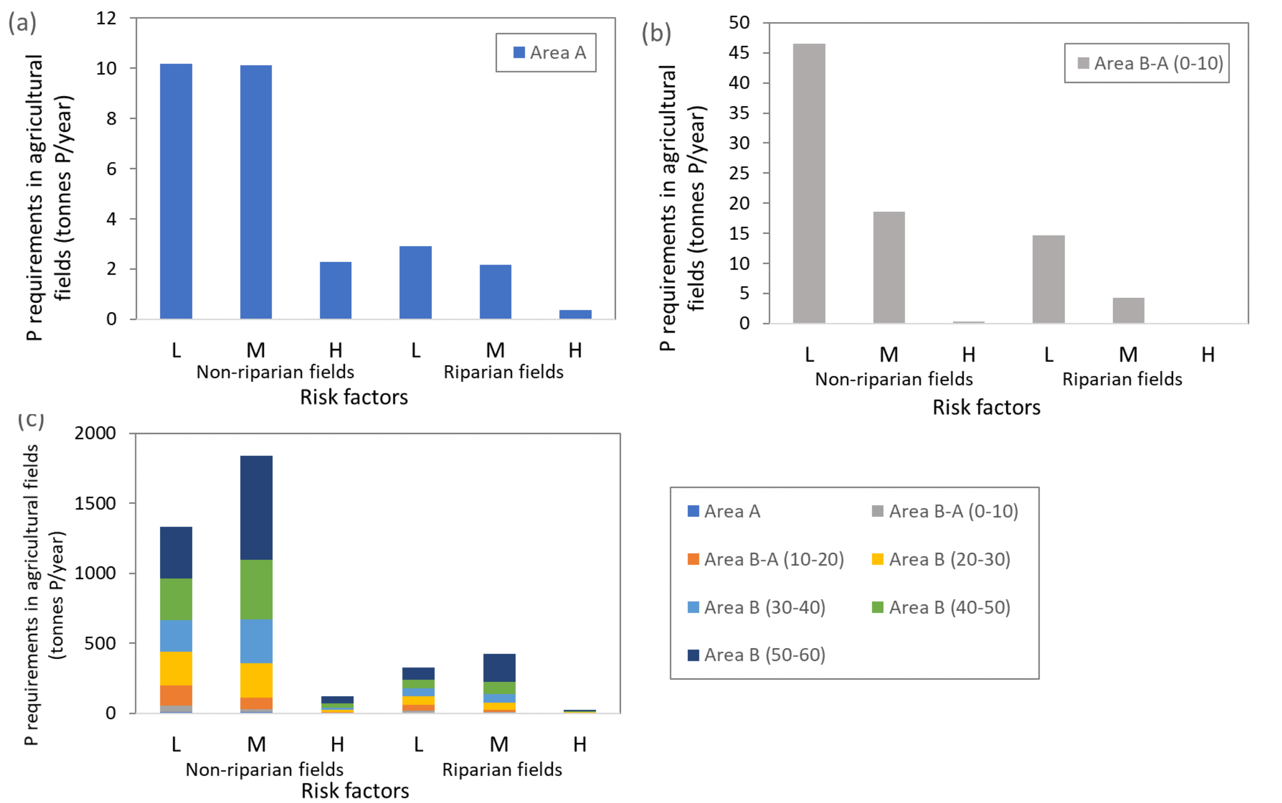


Figure 11. The sum of crop P requirements in fields according to the examined risk factors (riparian vs non-riparian fields and low, medium, high erosion risk). (a) shows the distribution of risk between the smaller land bank of the WFD waterbody. (b) shows the 0-10 km transport band distance of area B in which the recycled materials in the scenarios were able to be fully utilised (see e.g. Table 20) and (c) includes all the extended areas (up to the 60 km transport distance bands that denote the maximum distance of subsidised transport for the AD material).

P fertiliser usage was to fields in the high erosion risk category. Fig. 10c shows the whole area considered out to the 60 km transport distance (used since AD had subsidised transport to this distance). For this greatly extended area the potential P fertiliser usage denoted by the studied year of cropping returns shows that P use was dominantly to non-riparian fields and that high-risk fields were limited.

5.4.2. Potential interactions of risk factors between chemical fertiliser and differing replacement materials

The overall assessment of the field risks in the area A and the area B up to the 10 km transport distance where the recycled P materials could be utilised in the considered scenarios confirms low to medium erosion and connectivity risks to watercourses. In addition, the phosphorus sorption capacity in the area was found to be dominantly high at the resolution of available mapping. Therefore, the area of the scenarios of P replacement have relatively uniform risk factors for P fertilisers where these are applied under best management practices. The evidence base for differences in the relative risks of loss pathways of P leaching and potential P erosion with soil particles is not sufficiently developed to discriminate between relative risks of changing conventional chemical fertiliser for recycled materials such as AD, or sludges. This is a field of developing research and with need for further guidance. Hence, it is not possible at the present time to develop modelled outcomes for water quality differences between the scenario of conventional chemical fertiliser plus on farm manure and slurry usage and scenarios of replacement of chemical fertiliser with AD, sludges or struvite.

6. Conclusions and recommendations

Because we initially focussed on waterbodies failing WFD status for P pollution this included areas where larger urban wastewater P discharges occurred. As expected, this brought a strong spatial coupling between addressing failures in surface water quality of rivers and the urban populations where human-derived (food chain) P resource flows are magnified. Again, this is a known issue in terms of making so called 'linear P flows' adapted to 'circular processes'.

Two factors have become apparent through the current analysis. Firstly, that areas of population are centres of production of P-bearing waste materials but are not necessarily surrounded with immediate areas of agriculture with high P usage demands, hence the need to look at wider land bank areas that spread across catchments used in waterbody management. Secondly, local economic factors (job creation, policy for recycling etc) mean that active schemes for P recycling coincide in that space (such as food waste collection promoting AD generation simultaneous to effluent P processing centres) such that multiple nodes of collection and processing of P-bearing resources coincide.

Once the P bearing materials have exceeded the potential for agricultural production usage within transport spheres based on value for replacing chemical fertiliser, then economic incentives are required to distribute materials. This is caused in part by the need to find a market (even a subsidised one) to offset the industry having waste disposal charges. Within the area examined we found a centralised AD plant that subsidised the by-products of the energy generation for distribution to the local agricultural sector up to 60 km distance (via free transport to farms included in a relatively small gate price that was a third the cost of chemical fertiliser).

Due to several factors the P-bearing material associated with the WWTW could not be easily utilised in the local waterbody. These factors were the restrictions of usage on certain crops, the modelled generation of manure and slurry on farms (that we preferentially utilised before conventional fertiliser or any recycled materials) and the small area of the WFD defined waterbody. In the expanded area up to 20 km transport distance a deficit of available manures and slurry relative to the appropriate land bank allowed usage of the conventional sludge and this became the cheapest scenario, with a saving of £165K on overall area P fertiliser when compared to the use of other materials. This was due to the very low assumed sludge gate price of £2/kgP and the relatively low handling costs for this P dense material. However, this usage did not directly restrict P effluent discharges to the local watercourses. However, local sale of this material (as opposed to an expected net cost of removal as waste) may subsidise future improvements at the WWTW with subsequent effluent quality benefits. Despite the favourable economics of using the sludge because of its assumed conventionally processed current form (as estimated in the absence of available information) it has strong limitations in the classes of crops it can be applied to that limit its usage.

To evaluate a comparative material from the WWTW sector we undertook a hypothetical scenario of generation of struvite from Dunnswood WWTW. Such a material requires investment to produce and, to our knowledge, is not being currently generated at commercial scales in Scotland or utilised as part of the current P resource flows in agriculture. However, such a material is processed to the extent that stakeholder perceptions and regulatory barriers should be readily overcome if it were available resulting in wide usage possibilities for crop types. In our struvite scenarios the full mass of material was close to being able to be used when the local WFD waterbody area was considered and was readily utilised in the extended area within short transport distances. Cost savings in the overall fertiliser costs for the WFD waterbody area when the struvite is utilised on the land bank suggest that ~£20K annually could subsidise costs of production at the WWTW.

The distribution of available P resources in this area differed to the average national distribution of P resource flows to Scotland's agricultural soils (depicted in Box 1, section 2) of 20.4, 9.1, 0.3 and 1.3 ktonnes P annually of chemical fertiliser, animal manure/slurry, AD and sewage sludge, respectively (66%, 29%, 1% and 4% split of those four P fertiliser materials). The available resource of 40000 kgP of AD and 14053 kgP of sludge shows that this area has a large contribution from AD associated with the location of the AD plant at Dunnswood and that this was disproportionate to the usage of AD in agriculture at a national scale. The reference conventional fertiliser scenarios showed that the WFD waterbody area had 60% of P requirements from chemical fertiliser and 40% from manure/slurry with excess manure/slurry production for export to wider areas. The recycled scenario for this restricted area of the WFD waterbody showed that up to half the 40000 kgP resource of AD material could be utilised entirely replacing the chemical fertiliser due to usage against a wide range of crops due to its PAS accredited status. But that the conventionally treated sludge could not be utilised since its land bank was in competition with that of the already excess amounts of manure and slurry. In the extended area the material level 1 scenario with AD fully utilised within a transport range of 20 km road distance the fertiliser distribution was 56% manure/slurry, 34% chemical fertiliser and 10% AD. When the conventionally treated sludge was utilised in the extended area level 2 scenario this became 3% of the overall P fertiliser input with a corresponding reduction in chemical fertiliser. Compared to the national average P fertiliser material budgets the effects of local source availability have strong effects on the potential for distribution of P fertiliser types.

Within the WFD waterbody area the ability to use half of the available AD resource increased the overall area P fertiliser price by £8.5K. Considering the larger P demand of the 0-20 km transport distance area expanded area including areas outside of WFD waterbody the additional cost of AD use above conventional fertiliser was £21K. This is due to the greater spreading price for AD versus the conventional fertiliser even factoring in the subsidised transport within the AD gate price. The AD is a disproportionate influence on potential fertiliser components in this area. We know that the 40000 kgP annually of this resource is being commonly utilised in the local area up to the 60 km subsidised transport distance. The scenarios here show that this could be utilised more effectively in a smaller road transport distance of ~10 km if a significant number of farmers were persuaded to use it as chemical fertiliser replacement. This would limit some fuel use in the current distribution associated with the AD material but that persuasion is not leveraged by the current economic situation of AD pricing. It may be more perceptions of AD that need to be altered. However, we used a relatively high spreading price for the AD of £9/kg that makes the combined gate and handling costs of AD similar to chemical fertiliser (in fact slightly greater) and more specialised equipment is required to handle and spread the AD than for simple chemical fertiliser pellets.

In terms of assessment of constraints of material usage on soils we used literature based representative metal compositions for biosolids and struvite and the PAS accreditation thresholds for AD to calculate the maximum annual metal loadings permissible to soils under the Sludge Regulations in Agriculture 1989 (amended). The strictest limits were associated with potential for zinc and copper in biosolids and AD and for mercury in struvite. However, when these were converted to equivalent maximum P loading the very large thresholds 474, 860 and 10495 kgP/ha/year for biosolids, AD and struvite were excessive compared to field application rates intended to offset crop uptake (instead being relevant only when considering extreme cases of material loadings to 'sacrificial land'). Hence, under agricultural planning scenarios that match crop P usage, metal annual application rate limits are not likely exceeded. In practice this maximum average annual application rate is only one of two components of the requirements and additionally site-specific soil testing would set limits to maximum permissible metal concentrations in-situ; these data were not available to this study.

Currently poor knowledge exists on quantified and relative risks of P mobilisation from recycled P-bearing materials in soils. An example is the inadequate information on the P availability (and same for K availability) to the crop in the season after use and in following years. This is only considered to be well-studied for cases of farm-derived organic fertilisers manure and slurry, not for non-farm and industrial P bearing materials such as sludges and AD. Additionally, materials such as AD are subject to different processing (wet, dry fractionations) that result in different compositions with implications for mineralisation and availability of nutrients such as N and P to crops versus risks for diffuse pollution. This is a developing area of knowledge that requires extrapolation from current laboratory simulations to experiments at pot and field scales (much as sewage sludge to land research developed a decade ago). There is indication that fractions of AD can maintain highly mobile P in soil waters that may be a risk for P leaching, necessitating frameworks for field risks based on scientific knowledge. Likely the risks can be mitigated by good practice in P loadings and timing of applications generally, coupled with restrictions on certain fields. In the absence of a more rigorous assessment framework in this current report diffuse pollution risks were assessed using a very simplified procedure that defined areas of fertiliser replacement according to risk tiers associated with (i) proximity to watercourses (riparian vs non-riparian fields), (ii) erosion risk and the soil P sorption capacity translated from the relatively coarse mapping unit of soil Associations (mineralogy of parent materials). The assessment made here of risk factors however showed that the field areas dominating the potential P requirements were approximately 80% non-riparian (ie with a low connectivity risk), unlikely to be in a high erosion risk category (<10%) and the whole area was dominated by soils with a high P sorption capacity. Hence, in this region the field factors seem to indicate low prevalence of diffuse pollution risk factors for P prevails. The most important factors of capping P loadings to crop requirements (not allowing risky disposal of material in excessive amounts to a 'sacrificial' land area) and identifying isolated high-risk landscape-soil situations seems to be well carried out already under the waste licencing exemption processes. These seem thorough since they match site assessment to material and soil testing. It should be noted that when using alternative fertiliser materials under the system of paragraph 7 exemption there is no control on timing of material application to ensure it's applied at times of crop need, other than restrictions for times of frozen or waterlogged soils within the one-year licence period; hence timing of applications to crop requirements is in control of the farmer.

In summary, there is a potential to offset significant amounts of chemical fertiliser usage in Scottish agriculture in this scenario of local usage of P bearing materials to a land bank up to 20 km transport distance. The economics of this seem favourable and potentially familiarity with handling current pelleted chemical fertilisers and lack of familiarity with alternatives may be an issue in more widespread usage. However, in this area the occurrence of a relatively large resource of AD, coupled with some other constraints of usage meant that there was limited incentivisation to effectively utilise raw biosolid products from wastewater treatment. The exception to this is materials such as struvite from higher-level processing of wastewater materials (a hypothetical scenario in this case study since struvite is not generated in Scotland at present). If, as this study suggests, struvite can be produced and transported readily to wider land bank areas then this adds to the attraction of such materials alongside their lack of constraints in agricultural usage. The existing conventionally treated sludges had constraints of usage for many crop types and it may take a switch in infrastructure to enable alternative fertiliser materials such as struvite to be produced in order to couple the benefits of fertiliser replacement in agriculture and reductions in the P currently being lost in water waste streams via effluents. Methods developed here of field crop classes and pollution risk factors considered as a spatial field model linked from production nodes by road transport routing distances may provide a basis for spatial planning frameworks to encourage P recycling. It is most important that the potential recycled P resources are distributed to supplement or replace chemical P fertiliser using planning frameworks as opposed to being overapplied to more limited land areas. Further research is required on the crop availability and pollution risks for the growing varieties and masses of availabilities of P-bearing materials (for example fractions of AD across differing feedstocks) on which to base future agronomic guidance.

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APPENDICES

Appendix I.1. PR reserves: Estimation, distribution and implications

The ***Peak P model*** predicted that PR production would peak in 2030-2035 followed by depletion by 2100 (Cordell et al.,2009; 2011). Furthermore, that future shortage of P would threaten global food security, bioenergy production, and alter ecosystem structure and function due to the resulting stoichiometric imbalance in the stocks and utilisation of nitrogen and carbon (Neset and Cordell 2012; Penuelas et al.,2013). The model was initially adopted by the European Commission to support listing of PR as a "Critically Raw Material" (e.g. COM(2014)297 now withdrawn and replaced by COM/2017/0490). The Peak P model has received strong criticism mainly because it disregards a wide range of processes such as the interplay between PR demand and supply; the influence of technological advances in PR exploration, mining and precision agriculture; and the increasing effect of changes in dietary patterns and P in food waste (e.g. Clift and Shaw 2012; Giraud 2012; Daneshgar et al.,2018; Scholtz and Wellmer 2019). In a more positive light, Peak P can be shifted by the increasing use of recycled (secondary P) wastes (Geissler et al.,2018; Scholz and Wellmer 2019) and thus the Peak P model can be used as an early warning indicator to implement in advance adequate resource management measures towards a circular economy of P (Calvo et al.,2017).

The ***Lifetime of available reserves model*** assumes that the rate of P consumption will regulate the rate of PR reserve depletion, whereby a ratio of reserves to consumption can be applied to estimate the lifetime of available reserves. Market prices, production costs and technological innovations are dynamic and have significant influence on whether deposits can be deemed economic to exploit (USGS 2013). This means that reserve estimates must be frequently revised. Hence, van Kauwenbergh (2010) concluded that "there is no indication that phosphate production will peak in the next 20-25 years or even within the next century". The US Geological Survey (USGS) have adopted this approach PR reserves concluding that "There are no imminent shortages of phosphate rock" (USGS 2019).

The increasing demand for P fertilisers led to an unprecedent increase in the annual production rates of PR-based (manufactured) fertiliser production from approximately 120 Mt in the early 1990s to rates exceeding 220 Mt per year since mid-2000s (Daneshgar et al.,2018; USGS 2019). World consumption of phosphate (P_2O_5) contained in phosphoric acid, fertilizers and other uses is projected to increase to 50.5 million tons in 2022 from 47.0 million tons in 2018; Africa, India, and South America are predicted to account for about 75% of the projected growth (USGS 2019). The percent (%) PR supply to the EU in 2010-14 was sourced from Morocco (28%), Russia (16%), Syria (11%), Algeria (10%) and Finland (12%) (COM/2017/0490).

There are two major problems facing global PR reserves:

(i). Known PR deposits and reserves are unevenly distributed around the world in terms of both quantity and quality. Approximately 70% of global reserves occur in Morocco and the Western Sahara (USGS 2019). Therefore, many countries, including EU Member States, rely on PR imports for food production from a few supply countries, with potential geopolitical and public health implications. Except for Finland, Europe has no significant PR mines. Therefore, food production in the EU is dependent on imported PR (De Ridder et al.,2012; Schoumans et al.,2015). In terms of global PR production, Morocco and the Western Sahara and Russia account for only 12% and 5%, respectively, whereas China accounts for 52% of global production (USGS 2019).

(ii) The quality of PR reserves is declining, especially with respect to toxic metal pollution. The contribution of hazardous elements in PR ore depends on the geographic origin of the ore (Steiner et al.,2015). This adds to the complexity of estimating the marketability of PR reserves and setting regulatory limits for potentially toxic elements in manufactured P fertilisers. For example, phosphorites are known to contain hazardous elements, e.g. cadmium (Cd), chromium (Cr), mercury (Hg), lead (Pb), uranium (U), thorium (Th) and radium (Ra) (Mortvedt and Sikora, 1992; Kpomblekou and Tabatabai, 1994). Sedimentary ores from Morocco and Tunisia range up to 51 and 56 mg Cd/kg P_2O_5,respectively (Van Kauwenbergh 2001; Roberts 2014;). Brazilian PR may contain up to 182–220 mg U/kg. (Schmidt et al.,2011 cited in Geissler et al.,2018).

In terms of resource security, robust geopolitical risk scenarios (e.g. political instability, war and change in environmental or agricultural policies) for each country dependent on PR imports from each supply country are largely lacking. Countries without their own PR sources are vulnerable to fertiliser price increases associated with increases in export prices or tariffs from producing countries (Scholz and Wellmer 2019). For example, P fertiliser prices rose from 50 US$ per tonne in January 2007 to 350 US$ per tonne in 2008 as a result of changes in the fertilizer market policies in India coupled with falling value of the US dollar, rising transportation costs, fertilizer demand for biofuels production and China's introduction of a 100% fertiliser export tax (Khabarov and Obersteimer 2017).

Appendix I.2. The Circular Economy Package and Fertiliser Regulations under the Circular Economy Action Plan European Parliament 2018).

The Circular Economy Package (European Parliament 2018) sets legally binding recycling and waste reduction targets and more stringent rules on waste management to address the waste hierarchy:

- Prevention
- Preparing for reuse
- Recycling
- Other recovery
- Disposal

The Package amends four existing EU Directives. The new Directives in the Package are:

- Directive (EU) 2018/851 amending the Waste Framework Directive (2008/98/EC);
- Directive (EU) 2018/850 amending the Landfill Directive (1999/31/EC);
- Directive (EU) 2018/852 amending the Packaging and packaging waste Directive (94/62/EC);
- Directive (EU) 2018/849 amending the Directives: End-of-life vehicle (2000/53/EC); Batteries and accumulators (2006/66/EC); and waste electrical and electronic equipment (WEE) (2012/19/EU).

The Action Plan encourages the sustainable use of organic waste in agriculture to reduce the need of PR-based fertilisers. However, the use of recycled fertilisers is currently hampered by the fact that rules as well as quality and environmental standards differ across Member States. To address this situation, the Commission's Circular Economy Action Plan (COM/2015/0614) proposes a revision of the EU regulations on fertilisers amending Regulation (EC) No. 2003/2003. This will involve new measures to facilitate the EU-wide recognition of organic and waste-based fertilisers, thus stimulating the sustainable development of an EU-wide market for recycled-P fertilisers.

Appendix I.3. Scotland's initiatives enabling P recycling and recovery from waste streams

Scotland's first Zero Waste Plan (Scottish Government 2010) promotes zero-waste approaches alongside key provisions set out in the Scottish Government's Climate Change Delivery Plan (Scottish Government 2009) and the Climate Change (Scotland) Act 2009 linked to GHG reductions. The Plan is a vision for a Scotland where resource use is minimised, valuable resources (including P but not particularly referring to P) are not disposed of in landfills, and most waste is sorted into separate streams for reprocessing, leaving only limited amounts of waste to go to residual waste treatment, including processing to produce energy from waste facilities.

The ***"Making Things Last" strategy*** (Scottish Government 2016) builds on the Zero Waste Plan to maximise the value of bio-waste, in particular food waste and waste from the beer and whisky production sectors. Areas of ongoing activity refer to:

- Mapping bioresources, including P. For example, in 2015 the Scottish Industrial Biotechnology Development Group published the 'Biorefining Roadmap for Scotland' which laid down requirements for mapping of the wastes, by-products and agricultural residues which could potentially be used as bio-based feeds for biorefinery technologies (e.g. AD facilities) in Scotland.
- Investigating the potential for local biorefining hubs. A report on the "Biorefining Potential for Scotland" published by Zero Waste Scotland (2017) provided the first thorough map of bioresource arisings across Scotland and highlighted that there are at least 27 million tonnes of biomass (including by-products from the whisky-making industry, fruit and vegetable waste, mixed food waste, garden waste, and sewage sludge; see also Table 2-Section 2) available which could potentially be used as feedstocks in biorefining.
- Investing in Anaerobic Digestion. A study by ClimateExchange (CxC) conducted in Scotland found that without addition of other feedstocks, the anaerobic digestion (AD) of slurry and farmyard manure (FYM) has a proven poor business case at both farm and centralised facility scales, with a high capital cost, low energy yield and absence of gate fee (Ford et al., 2017). Slurry and FYM arise in very high volumes across all areas of Scotland, the greatest arisings of slurry and FYM observed in Dumfries and Galloway and Aberdeenshire. The study suggested exploring opportunities for co-digestion of slurry/FYM from these areas with other types of feedstocks that are available within a reasonable distance.

Appendix II.1. Global natural and anthropogenic P cycles

Natural P cycle (Ruttenberg 2003; Filippelli 2008)

Phosphorus (P) enters soils via tectonic uplift of phosphorus-bearing rocks and their subsequent exposure to the forces of weathering (i.e. dissolving of rocks and minerals on the surface of the Earth) and physical erosion. Subsequently, soil P is subjected to complex biogeochemical transformations determining P pools and P availability for terrestrial plant uptake. Soil P (dissolved and particulate) is transported to lakes and oceans by means of surface and subsurface runoff and leaching, and, once deposited in sediments, is lost from the system until the cycle begins anew after 10 to 100 million years with uplift of sediments into the weathering regime.

Anthropogenic P cycle

In addition to P inputs to soils and waters through the global natural P cycle, anthropogenic P (MacDonald et al.,2016) is released to the environment from human actions or management, including application of PR-derived fertilisers and their subsequent redistribution via recycling of manures and human wastes and soil erosion.

The anthropogenic P cycle has disrupted the global natural P cycle in many ways

- Anthropogenically-affected P flows to soils greatly exceed natural flow, 29 and 10 Mt/year, respectively (Smil 2002).
- The anthropogenic P cycle is dominated by trade of fertilizer and food, management of food waste and sewage. In contrast, the natural P cycle is dominated by fluvial fluxes and processes (Powers et al.,2016).
- Local recycling of any given P molecule occurs once in an anthropogenically-affected P cycle compared to 47 times under a natural P cycling state, prior to losses to waters (Daneshgar et al.,2018) related to:
- Intensifying anthropogenic P removal through crop harvesting post-1945 (Steffen et al.,2015).
- Inefficient agro-practice (e.g. failure to return P in crops and manure in the arable land and grassland where it was produced) (Gillingham et al.,1980; Syers et al.,2006).
- Poor soil management resulting in P accumulation in soils and P losses through erosion and leaching.
- Global societal P use inefficiency in food-related systems, whereby P inputs for food production are lost as food waste, animal waste and sewage (Daneshgar et al.,2018). In Europe, it takes 4kg of PR-derived fertiliser to produce 1kg P as food, with over 40% of all surplus inputs ending up in the soil and over 50% lost from the system, of which 27% is to waterbodies (van Dijk et al.,2016).

Appendix II.2. Summary of key soil biogeochemical transformations determining P pools and P plant availability.

Key soil biogeochemical transformations determining P pools and P plant availability include:

- Uptake by terrestrial plants, whereby inorganic P, mainly orthophosphate (H2PO4- and H2PO42-), solubilised during weathering or added as soluble PR-derived fertiliser, is available for uptake by terrestrial plants.
- Mineralisation of non-living organic P, which returns P in plant and animal biomass to the soil solution by slow microbial decay mainly of litterfall, crop residue and animal faeces, which can then become available for plant uptake.
- Precipitation, whereby inorganic P reacts with dissolved iron, aluminium, manganese (in acidic soils), or calcium (in alkaline soils) to form phosphate minerals, which are not available to plants.
- Adsorption, i.e. binding of inorganic P on soil clay minerals decreasing its availability for plant uptake.
- Microbial immobilisation, whereby soil microbes turn inorganic P into microbial P, which is not available plant uptake.
- Microbial re-mineralisation of immobilised P, i.e. release of microbially bound inorganic P.

Appendix III.1. Legislation on P recycling

Table III.1. EU-level policies addressing P-use and control through the setting of P limits in the water environment, discharges, agricultural use of P-containing fertilisers and wastes and limits on microbiological and chemical composition of P-containing fertilisers.

Table III.1a. EU-level policies on water protection

Legislation	Provisions	Objectives
Water Framework Directive (2000/60/IEC)	Water quality -Stipulates P standard in surface and groundwater waterbodies	Achieve and maintain a good ecological status (GES) and chemical status for all surface waters and groundwaters, respectively by 2027.
Urban Wastewater Directive	Water quality - Prescribes limits in P discharges from wastewater treatment works (WWTW)	Protect the environment from adverse effects of wastewater discharges from cities and certain industrial sectors
Marine Strategy Framework Directive (2008/56/EC)	Marine industry -Boundaries for P in transitional and marine waters	Achieve (GES) of the EU's marine waters by 2020 and to protect the resource base upon which marine-related economic and social activities depend
Regulation (EU) 259/2012 amending Regulation (EC) 648/2004		Lay down requirements for the use of phosphates and other phosphorus compounds in consumer laundry detergents and consumer automatic dishwasher detergents

Table III.1b. EU-level policies on agricultural management

Legislation	Provisions	Objectives
Common Agricultural Policy (CAP) (European Commission n.d.)	-Requires cross-compliance for receiving Pillar I payments -Allows for additional payment through the implementation of Pillar II agri-environment schemes	-Keep land in good agricultural and environmental condition. -Code of Good Farming Practice prescribes rules e.g. manure spread, treatment and storage, and application -Prescribes schemes for protecting or improving water quality, e.g. fertilizer application restriction/organic farming
Nitrates Directive (91/676/EEC)	-Stipulates maximum amounts of nitrogen in manures applied on land (i.e. 170kgN/ha/y), which can indirectly reduce the amount of P added through these fertilisers.	Improve water quality by protecting water against pollution caused by nitrates in animal manures, chemical nitrogen fertilisers and other nitrogen-containing materials spread onto agricultural land.

Table III.1c. EU-level policies on waste management and soil protection

Legislation	Provisions	Objectives
Council Directive 86/278/EEC (as amended) (aka Sewage sludge Directive)	-Requires sludge to be treated before use in farming. -Sets standards for seven heavy metals in sewage sludge and amended soils -Restricts sludge use on grassland or forage crops that are going to be grazed and for at least three weeks before harvest. -Bans sludge use on fruit/vegetables during the growing season, and on crops eaten raw for 10 months before and during the harvest. -Requires national authorities to keep records of sludge amount, composition, properties, and production and use places	Set rules on how farmers can use sewage sludge as a fertiliser, to prevent it harming the environment and human health, by compromising the quality of the soil or surface and ground water.
Landfill Directive (1999/31/EC) (as amended)	Regarding bio-waste (as a potential P source)^: - Requires amount of municipal bio-waste to landfills to be reduced to 35% of 1995 levels by 2016 -2020 -Bans disposal of biodegradable household waste to landfill after 2030.	Prevent or reduce as far as possible negative effects on the environment, in particular the pollution of surface water, groundwater, soil and air, and on the global environment, including the greenhouse effect, as well as any resulting risk to human health, from the landfilling of waste, during the whole life-cycle of the landfill
Waste Framework Directive (2008/98/EC) (as amended)	-Prohibits mixing of hazardous substances with biodegradable municipal waste -Requires a minimum of 55% by weight of municipal waste by 2025, 60% by 2030, and 65% by 2035 to be prepared for re-use after its collection and recycling -Identifies "Land treatment resulting in benefit to agriculture or ecological improvement" as a potential waste recovery option	Protect the environment and human health by preventing or reducing the generation of waste, the adverse impacts of the generation and management of waste and by reducing overall impacts of resource use and improving the efficiency of such use, which are crucial for the transition to a circular economy and for guaranteeing the Union's long-term competitiveness
Animal By-products Regulation (EC No 1069/2009	-Allows animal by-products not posing a risk to human health to be used as organic fertilisers and soil improvers -Allows digestion residues from transformation into biogas or compost to be placed on the market and used as organic fertilisers or soil improvers	-To lay down public and animal health rules for animal by-products and derived products in order to prevent and minimise risks to public and animal health arising from those products and to protect the safety of the food and feed chain.

^ Bio-waste is defined as biodegradable (i.e. that is capable of undergoing anaerobic or aerobic decomposition), and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants. It does not include forestry or agricultural residues, manure, sewage sludge, or other biodegradable waste such as natural textiles, paper or processed wood. It also excludes those by-products of food production that never become waste.

Table III.1d. EU-level policies on P resource/trading safety

Legislation	Provisions	Objectives
List of Critical Raw Materials for the EU (European Commission 2017)	PR is listed as a critical raw material for the EU (see also Section 2.1)	Identify the raw materials with a high supply-risk and a high economic importance to which reliable and unhindered access is a concern for European industry and value chains.
Regulation on organic production and labelling of organic products (EC No 834/2007)	Regarding organic fertilisers: -Sets maximum concentrations in mg/kg of dry matter in composted or fermented mixture of household waste.	Lay down the requirements on organic production and labelling of organic products regarding organic production, labelling and control
New European Fertiliser Regulation by 2022 (awaiting final approval) (European Parliament 2019)	Revises Regulation (EC) No 2003/2003 (Fertiliser Regulation) by: -including all types of fertilisers (mineral, organic, bio stimulants, growing matters, industry by-products, etc.) -Setting harmonised limits for all contaminants of concern to public health. -Prescribing a voluntary low Cd label in P fertilisers (Cd < 20 mg Cd/kg) in addition to the statutory limit of 60 mg Cd /kg of P fertiliser**. - promoting increased use of recycled materials for producing fertilisers enhancing circularisation - easing market access for innovative, organic fertilisers - establishes EU-wide quality, safety and environmental criteria for "EU" fertilisers (i.e. those which can be traded in the whole EU single market	Guarantee the functioning of the internal market while ensuring that EU fertilising products on the market fulfil the requirements providing for a high level of protection of human, animal, and plant health, of safety and of the environment.

**Average Cd content in EU fertilisers is around 34 mg/kg of phosphate and ~8% of the EU's phosphate fertilisers would be in breach of the 60 mg limit (Financial Times - FT 2018). Russia PR exports are naturally low in Cd whereas Moroccan PR, which largely contribute to western Europe P fertiliser imports are naturally high in Cd.

Appendix III.2. Scottish regulations and policies underpinning P recycling.

Water protection

- The Water Environment and Water Services (Scotland) Act 2003 (as amended), also reported as the WEWS Act, which transposes the Water Framework Directive (WFD) (2000/60/EC) to national law. Under the WEWS Act SEPA must assess and address the pressures impacting the water quality in rivers and coastal through the development of six-yearly River Basin Management Plans (RBMP) with the aim to achieve good classification status (or, if this is not possible, to reduce pressures) by 2027. Regulations for monitoring and P standards in freshwater systems to inform status classification and the RBMP process are also in place.
- The Water Environment (Controlled Activities) (Scotland) Regulations 2011 (as amended), also known as CAR, which was developed to help SEPA implement the WEWS Act and support the RBMP process. Under CAR, SEPA controls rural diffuse pollution and direct (point-source) effluent discharges to the water environment. The General Binding Rules (GBR) specified in CAR describe compliance with activities posing minimal or low risk to water quality status and soil P status. Examples refer to rules for the: storage and application of chemical and organic fertilisers (GBR 18); keeping of livestock (GBR 19); and cultivation of land (GBR 20).
- The priority catchment approach was launched in 2011 to help SEPA prioritise action in delivering the objectives set under the RBMP process (DPMAG-SEPA 2017), e.g. implement waterbody-scale measures to promote compliance with the WFD-based P and phytoplankton standards.

Agricultural management

- EU-level legislation to prevent nutrient pollution from agriculture has mainly relied on the Nitrate Directive and the measures underpinning the implementation of the CAP, which do not explicitly address P. However, Denmark, Netherlands, Germany, and Italy, accommodate P explicitly in their national agricultural nutrient management regulations (Barreau et al.,2018).
- The code of good agricultural practice, giving practical advice to farmers and others on the Prevention of Environmental Pollution from Agricultural Activity (PEPFAA) (Scottish Executive 2005a).
- The Four Point Plan, which contains simple guidance on how to reduce dirty water around the farm, improve nutrient use, carry out a land risk assessment for slurry and manure and manage your water margins (Scottish Executive 2005b).

Waste management interactions with soil and agricultural protection

- In the UK, demands for public confidence in the safety of composts and digestates used as alternative fertiliser products to address the Landfill and Waste Management Directives led to the British Standards Institution's Publicly Available Standards (PAS) for composts (BSI 2011; SEPA 2017a; WRAP 2016)) and AD digestates (BSI 2014; WRAP 2016) The PAS criteria reference global initiatives on standards for composts and AD digestates that are achievable and safe.
- To improve and standardise application of sewage sludge to agricultural land on a UK level, an agreement, known as The Safe Sludge Matrix (ADAS 2011), was made between Water UK representing the 14 UK Water and Sewage Operators and the British Retail Consortium (BRC) representing the major retailers, which came into force on 31 December 1998. It consists of a table of crop types, together with clear guidance on the minimum acceptable level of treatment for any sewage sludge (aka biosolids) based product which may be applied to that crop or rotation (Figure 3, Section 2.3.6). The Safe Sludge Matrix was originally intended to be incorporated into the Sludge (Use in Agriculture) Regulations 1989 (as amended) and into the UK Code of Practice for Agricultural Use of Sewage Sludge (DEFRA 2018)[14]. As of 2019 (and at the time of writing this Section), The Safe Sludge Matrix remains a UK-level voluntary agreement.
- With respect to manure and slurry use as fertilisers, it is important to remember that the recommendation from the Food Standards Agency (FSA) is that manure should be stacked for 8 weeks to reduce the risk of spreading antibiotic-resistant bacteria (FSA 2009). Where manure is to be applied to land before growing ready-to-eat crops such as salad leaves, the FSA (2009) recommend that manure should be stored for at least 6 months prior to use to kill microbial pathogens, with no fresh additions being made to the store during this period. FSA recommendations are comparable to the provisions of the Safe Sludge Matrix.
- The Waste Management Licensing (Scotland) Regulations 2011 requires SEPA to assess registration forms:
  - o For the application of organic materials specified as wastes to land (agricultural or else) and to

14 This applies for England, Wales and Northern Ireland. For Scotland see SEPA 2020.

allow it as an exempt activity if the waste applied provides agricultural or ecological benefits (Paragraph 7 of Schedule 1 and Schedule 2); see also Box III.2.1. Paragraph 7 exemptions also apply for spreading waste on non-agricultural land for ecological improvement. Applications to SEPA must include a certificate from an appropriate technical or professional expert describing how the waste application will result in benefit to agricultural or ecological improvement. Waste applied under Paragraph 7 exemption must be analysed in relation to physio-chemical parameters (e.g. % dry solids content, pH, conductivity, readily plant available ammonium-nitrogen, BOD and COD, and C/N ratio) as well as microbiology, heavy metal content and other substances, depending on type of waste (see also Box III.2.1 and Table V.3.2). As noted in section 2.3.6 AD derived entirely from farm waste or crops grown entirely for AD production purposes can be spread anywhere without para 7 or PAS110. Other Regs and code of practise continue to apply, e.g. PEPFAA, 4 point plan, CAR GBRs, NVZ Regs etc.

o For the storage and spreading of biosolids on non-agricultural land (Paragraph 8-Schedule 1).

Heavy metal concentrations in biosolids and the soil before amendment are compared against the mandatory limits set in The Sludge (Use in Agriculture) Regulations 1989 (as amended)[15].

- The Animal By-Products (Miscellaneous Amendments) (Scotland) Regulations 2015 and the Animal By-Products (Enforcement) (Scotland) Regulations 2013, which transpose into Scots Law the respective EU Regulations, stipulate that pasture land cannot be used for grazing within 2 months (for pigs) and 3 weeks (for other farmed animals) of applying materials derived from animal by-products (ABP). Farmers who use animal by-products must keep records of the date, quantity and description of the materials applied, and the date on which pigs and other farmed animals first have access to the land after application.

Table III.2.1 gives permissible levels of PTEs (in this case trace metals) in soil following application of sewage sludge under the Sludge (Use in Agriculture) Regulations 1989 (as amended), which applies in England, Wales and Northern Ireland

15 See also SEPA 2020.

Table III.2.1. Maximum concentrations of Potentially Toxic Elements (PTEs) in soils (0-25 cm depth) after application of sewage sludge and maximum annual permissible rates of addition under the Sludge (Use in Agriculture) Regulations 1989 (as amended).

Potentially Toxic Element (PTE)	Maximum permissible concentration of PTE in soil (mg/kg dry solids)				Maximum permissible average annual rate of PTE addition over a 10 year period (kg/ha)
	pH[3]	pH	pH	pH	
	5.0-5.5	5.5-6.0	6.0-7.0	>7.0	
Zinc (Zn)	200*	250*	300*	450*	15
Copper (Cu)	80 (130)	100 (70)	135 (225)	200	7.5
Nickel (Ni)	50 (80)	60 (100)	75 (125)	100	3
	for pH 5.0 and above				
Cadmium (Cd)	3				0.15
Lead (Pb)	300				15
Mercury (Hg)	1 (1.5)				0.1
Chromium[2] (Cr)	400 (600)				15
Molybdenum[2] (Mo)	4				0.2
Selenium[2] (Se)	3 (5)				0.15
Arsenic[2] (As)	50				0.7
Fluoride[2] (F)	500				20

Box III.2.1. SEPA Paragraph 7 exemption to SEPA Waste Management Licencing. Source: SEPA 2015.

- Should be used for <50 ha applications but can be different areas where multiple farms are considered together, <1250 tonnes storage
- Requires a recent (<12 months) analysis of soils (P status, Olsen P or Morgan P, pH, SOM, N, K, Mg, metal contaminants etc) but this can be extended to 36 months gap if additional information on all the fertilisers used since the last soil sampling is provided.
- Requires for analysis of waste material and variability (total P by Aqua Regia. C, N, K, Mg, metals, plastic) in the case of AD and compost.
- A location plan is needed showing watercourses, boreholes, wells etc
- The purposes are to benefit
    - o ecology (habitat change/creation),
    - o agriculture (acid neutralising, nutrients, soil moisture benefits, soil biota, soil texture and considerations such as land levelling (not allowed for organic-rich materials such as compost or AD))
- Other input required: application rate and style, current land use, crop and nutrient requirements

8. Purpose of the treatment	
(i) The purpose of the treatment is for: (Tick the relevant box and complete the corresponding section)	
ecological improvement (go to section 9 - ignore 10 & 11)	
Agricultural improvement (go direct to section 10)	

9. Ecological Improvement	
(i) You must include a document (described as a certificate in Schedule 3 to the Regulations) describing how the treatment will result in ecological improvement. This should identify individual species or habitat type and discuss how they will benefit from the treatment. Please note: failure to provide sufficient evidence to back the claim of ecological improvement will result in your notification being refused. The statement shall be prepared by or based on advice from a person with appropriate technical or professional expertise. The document should state this experience and be signed and dated by that person. Tick box if included.	
(ii) You must also include a document giving an assessment of the risk of pollution posed by the treatment and any preventative measures proposed. Tick box if included.	

10. Agricultural Benefit	
(i) Tick each correct statement in the list below:	
The waste will increase crop yield/quality	
The waste will improve the chemical properties of the soil (e.g. pH)	
The waste will improve the physical properties of the soil (e.g. tilth)	
The waste will improve the biological properties of the soil	
The waste will improve any soil moisture deficit	
The waste will have other benefits	
(ii) You must include a document (described as a certificate in the regulations) describing how the treatment will result in agricultural benefit. For each of the stated aims (10(i) above) you ticked, explain using the results of chemical analysis how the application of the specified waste will deliver agricultural benefit. You should detail optimum crop requirements, what the waste supplies and when the crop requires the benefit associated with the waste. Please note: failure to provide sufficient evidence to back the claim of agricultural benefit will result in your notification being refused. The statement shall be prepared by or based on advice from a person with appropriate technical or professional expertise. The document should state this experience and be signed and dated by that person. Tick box if included.	
(iii) You must also include a document giving an assessment of the risk of pollution posed by the treatment and any preventative measures proposed. Tick box if included.	

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11. Individual Field Assessment - A copy of this page should be completed for each field.			
Field Number/Identifier & size (Hectares - Ha) (as per your submitted map) (this number/identifier should also be on associated soil analysis)			Ha
(i) Current crop/use			
(ii) Planned crop/use			
(iii) Identify any wastes, including agricultural (eg manure or slurry), previously used to treat the land in the 6 months prior to the date of the intended treatment (include type of waste, approximate quantity, method of application)			
(iv) Please circle the most appropriate answer:			
Will the waste be spread in accordance with the PEPFAA* code **Prevention of Environmental Pollution from Agricultural Activity*			Y/N
Maximum field slope (any part of field)	<5°	5-10°	>15°
Proximity to watercourse/ditches (nearest part of field)		<15m	>15m
Proximity to spring, well or borehole (nearest part of field)		<50m	>50m
Proximity to drinking water supply (nearest part of field)		<250m	>250m
Proximity to dwelling (nearest part of field)		<50m	>50m
Average depth to water table	>1m	0.5-1m	<0.5m
Are buffer strips to be established in this field? *(mark on associated plan)*			Y/N
Have field drains been installed within the last year?			Y/N
Give the principal soil texture of the field, e.g. clay, sandy loam, silt by reference to the UK-ADAS soil textural triangle.			
OFFICE USE ONLY - FIELD ACCEPTABLE			**Yes / No**

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General resource management

- To promote the acceptance of recycling biosolids to agricultural land through a process of risk assessment, operational controls, third party audit and stakeholder reassurance, the UK Water Industry developed the Biosolids Assurance Scheme (BAS) Standard (UK BAS 2017). The BAS is based on the Sludge and Waste Management Directives and associated national regulations and best practice. It is operated and audited by Assured Biosolids Limited (ABL), a not-for-profit company, owned by the 11 mainland UK Water and Sewerage Companies. As of 2018, the scope of the Scheme includes the treatment of sludge (including source material risk assessments), transport, storage and application to agricultural land (UK BAS 2017). The Scheme divides Biosolids Source Materials into three categories: A, for domestic and industrial waste water; B, for septic tank material and water treatment sludge; and C, for feedstock material, e.g. typically, material of non-water industry origin (e.g. organic materials supplied by a third-party into Sludge Treatment Centres-STC), green waste, woodchip and sawdust.

Appendix IV. The fertiliser requirements of different crops and what forms are currently being applied

- Total usage of manufactured P_2O_5 fertiliser in Scotland was 48 kt for 2018 (British Survey of Fertiliser Practice - BSFP 2020). P_2O_5 use in Scotland has markedly declined from 81 kt in 1966 to 34 kt in 2009, which is the lowest use in the record due to high PR prices. Thereafter and until 2018, this decline has slowed, and total manufactured phosphate use has been more stable, ranging between 44– 54 kt. However, use is still approximately half that compared to use between 1965 and 1985, the implication being that residual phosphorus and manure fertilisers contribute to meeting crop requirements.
- In 2018, overall P_2O_5 application rates in Scotland were 27 kg/ha for tillage crops and 8 kg/ha for grassland (BSFP 2020). A breakdown of phosphate application rates per crop type and crop area in Scotland is presented in Table IV.1. Phosphate application rates declined markedly between mid-1990 and 2009, but thereafter there was some recovery and relative stability until 2018.

The existing recycling practice associated with utilisation of animal-derived manures and slurries in agriculture.

- The Sludge Review by the Scottish Government (2016) reports that approximately 50,000 Mt of animal manure and slurry are spread on land each year in Scotland.

The potential recycled-P resources from sewage.

- In 2017/18 the quantity of sludge generated in Scotland was 120, 032 tonne dry solids(tds), the majority of which came from the PFI assets – 106,292 tds – and Scottish Water's figure of 13,740 tds (Water Industry Journal - WIJ 2018).
- Approximately 70,000 tonnes of sewage sludge are spread each year on land in Scotland (Scottish Government 2016).

The potential for recycled-P resources as by-products from anaerobic digestion.

- Approximately 130,000 tonnes of compost and digestate are spread on land each year in Scotland (Scottish Government 2016).

Table IV.1. Overall manufactured phosphate (P_2O_5) fertiliser application rates per crop type (BSFP 2020).

Type of crop	Crop area receiving dressing[1] (%)	Overall phosphate application rate (kg P_2O_5/ha)
Winter wheat	88	58
Spring barley	92	48
Winter barley	90	56
Oats	78	42
Potatoes	95	125
Winter Oilseed rape	100	55
Other crops	48	19
All tillage	87	50
Grass <5 years old	59	19
Grass >5 years old	56	11
All grass	57	13
All crops and grass	68	26

1. The term dressing cover is used to describe the proportion of crop area treated with any dressing of the fertiliser nutrient in question and is stated as a percentage. It is included here because arithmetically, overall application rate is equivalent to the result of multiplying the average field rate of application (not shown) by the proportion of crop area that receives any nutrient dressing.

- The amount of food waste going into AD is only 125,000 tonnes of the 183,980 of municipal waste which is re-used, recycled or composted (RRC) (Zero Waste Scotland 2017).
- Data on the P fertiliser products of the Scottish AD industry are not easy to extract. Ford et al. (2017) report 46 Scottish AD facilities:
  - six facilities only process sewage sludge
  - eight process municipal/commercial waste
  - eight process industrial waste
  - 26 process waste or non-waste agricultural feedstocks, with three facilities recorded as slurry/FYM only facilities. Two of slurry/FYM only facilities are very small and were commissioned in 2016. One is medium sized (28,000 tonnes per annum) and processes poultry manure (higher gas yield) alongside cattle slurry.
  - 16 out of 46 facilities are reported to process animal slurries and manure alongside other wastes and crop feedstocks, with a combined processing capacity of 300,500 tonnes per annum.

Appendix V.1. P solubility and crop P availability

Manure - Slurry

- The Nutrient Management Guide RB209) by the Agriculture and Horticulture Development Board (AHDB 2019) updates the nutrient content of livestock manures. The Guide points out that nutrient content depends on a number of factors including the type of livestock and the diet and feeding systems, storage time, amount of bedding.
- Livestock manures contain a broad range of P-compounds with very different chemical structure and solubility (Table V.1), and therefore differing in plant availability. Based on the typical % P_2O_5 availability values, around 60% of P_2O_5 will be available to the next crop grown, with the remainder released slowly over the crop rotation (AHDB 2019).
- In manure, dominant organic P forms are phytic acid, phospholipids, DNA, pyrophosphates.
- Dominant inorganic P forms have relatively highly plant availability and include hydroxy apatite/octo-calcium, phosphate, P adsorbed to Al hydroxides, di-calcium phosphate, struvite, crystalline amorphous and aqueous phosphates (AHDB 2019; Kratz et al.,2019).

Recycled fertilisers (composts, anaerobic digestates, biosolids, struvite, ashes, biochars)

Specific P compositions

The majority of P in recycled-P fertilisers is in inorganic P forms:

- Composts and digestates: > 55% of Total P is inorganic. It must be noted that both composting and digestion lead to the transformation of organic P into inorganic P and reduce the ratio of easily soluble P to recalcitrant P compounds (Möller and Müller 2012).
- Biosolids: >85% of Total P is in inorganic form (Möller et al., 2018).
- ABP: the primary P mineral is Ca-deficient hydroxyapatite, which is more reactive than apatite PR (Möller et al.,2018).
- Struvite: the primary form is phosphate incorporated into a crystal with Mg^{+2} and another cation, with Ca-, Al-, or Fe- as co-precipitates (Möller et al., 2018; Katz et al., 2018)
- Ashes from sewage and organic waste: Normally, crystalline molecules such as calcium phosphate (aka Whitlockite) or similar compounds are present, but at incineration temperatures >700°C the formation of hydroxyapatite is also reported (Möller et al., 2018).
- In composts, digestates, and biosolids, the majority of organically-complexed P is in the form of orthophosphate monoesters. These can be of varying availability to plants and are often considered P storage compounds in plants and soils. The more soluble organic P forms such as diesters, phospholipids and nucleic acids often make up less than 5% of total P (Möller et al., 2018).

Table V.1. The P solubilities compared for conventional chemical fertilisers and phosphate-bearing rock, manures and recycled-P materials. The colour denotes low P solubility (blue cells), moderate (no shading) and high solubility (bold font, grey cells).

P material	P content	Solubility % of P total in different extractants (mean values)			
		Pwater	Pnac (neutral NH_4 citrate)	Pca (citric acid)	Pfa (formic acid)
Triple superphosphate (TSP)	45-46% P_2O_5	**88**	**90**	**95**	**100**
Di-ammonium phosphate (DAP)	46% P_2O_5	**97**	**93**	**94**	**99**
Mono-ammonium phosphate (MAP)	52% P_2O_5	**100**	**94**	**97**	**100**
Rock Phosphate-PR (e.g. Gafsa) -Tunisia	27-33% P_2O_5	<0.1	21	38	79
Ground PR -Morocco	13.38% of TP	<0.1	2	4	6
Dairy FYM		31	**75**	**66**	
Pig FYM		87	**92**		
Pig and cattle slurry		25	**91**	**85**	
Liquid AD-mixed waste		6-77	**70-86**	**47-83**	
Liquid AD-vegetable		19	**81**	**80**	
Solid AD		1	**74**	50	
Sewage sludge-EBPR		4	**94**	**94**	52
Sewage sludge-chemically precipitated			**57-100**	**7-82**	
Struvite		<1-7	**69-96**	**37-100**	**47-100**
ABP ash (untreated)		<0.1	15	50	48
Wood ash (untreated)		<0.1	**77**	**78**	
Cereal ash (untreated)		10	60	52	
Sewage sludge (untreated)		<1	10-56	30-55	28
Biochar -Manure		2	**70-82**	**66-89**	**72-94**
Biochar-ABP		<1-3	28	**72**	**81-99**
Biochar-sewage sludge-EBP		<1-7	**60-90**	43-97	36-68
Biochar-sewage sludge-chemically precipitated		<1-5	28-93	6-100	26-79

Refs: SRuC 2015; Kratz et al.,2019; Zapata and Roy 2004; Korzeniowska et al.,2013

Appendix V.2. General properties of P fertilisers

- DM: Digestates and biosolids that are not dewatered are characterised by low contents (2-10%), whereas composts have intermediate contents (50-75%) and ashes and dried Animal-by-Products (ABP) have high contents (>90%) (Möller et al., 2018).
- P concentrations: Composts contain the lowest concentrations (1.5-4.5 g/kg DM) followed by biosolids (2.5-14.5 g/kg DM), liquid digestates (4-31 g/kg DM), struvite (60-130 g/kg DM), ABP (60-165 g/kg DM) and ashes (40-189 g/kg DM). However, the P concentration of ash-based fertilisers depends on the source materials used (e.g. Nanzer et al.,2014) and the combustion temperature (Christel et al.,2014).
- Nitrogen Concentrations: Composts are characterised by low concentrations (7-20 g/kg DM) due to N losses during treatment and storage. However, high concentrations (>25 g/kg DM) are reported for urban organic waste digestate, ABP, biosolids and struvite (Möller et al., 2018).
- As a result of the above the derived N:P ratio is a key parameter in agronomic use.

Appendix V.3. Summary of contaminant content of chemical and recycled-P fertilisers

V.3.1. Chemical (PR-derived) fertilisers

The global sources of rock phosphate have themselves associated trace elements that are carried over into the chemical fertiliser products. Although the summary of data in Table V.3.1 is old this gives an idea of the contents. It is generally discussed that the contaminant situation is worsening for some elements especially Cd, as the purest sources are used and lower quality deposits are accessed and used; however data is not always forthcoming from mining companies. PR contains various metals as minor constituents in the ores. Varying amounts of these elements are transferred to P fertilizers in production processes, and later are applied to soils with these fertilizers. The PR-derived fertilizers, are historically of different origins in the western and eastern parts of the European continent and have a different composition in heavy metals. For example, while the Russian volcanic Kola PR, the main source of P fertilizers in Eastern Europe, is practically free of Cd, that from Morocco, the main source of P fertilizer in Western Europe and the UK, contains considerable amounts of Cd (Csillag et al., 2006).

V.3.2. Contaminants in materials available for P recycling

A summary of the considerations of contaminants in different materials added to soils is given in Table V.3.2. The table comes from the guidance given by SEPA as to the investigations (i.e. analysis or proof of the material compositions assessed against site suitability) that comes when permission is sought to add 'wastes' to land.
This takes the definitions of contaminants under three categories, namely: Microbiology such as pathogens; PTE, Potentially toxic Elements that generally describe the trace metals; Prescribed substances, Substances prescribed in Schedule 6 to The Environmental Protection (Prescribed Processes and Substances) Regulations 1991, i.e. P, organic solvents, pesticides, azides, halogens and their covalent compounds, metal carbonyls, organometallic compounds, oxidising agents, alkali metals and polychlorinated substances.

Table V.3.1. Average heavy metal concentrations in PR deposits and estimated inputs to soil by P fertilisers. Source: Kongshaug et al.,1992.

PR deposit	Heavy metal concentration, mg/kg						
	As	Cd	Cr	Pb	Hg	Ni	V
Russia (Kola)	1	0.1	13	3	0.01	2	100
USA	12	11	109	12	0.05	37	82
South Africa	6	0.2	1	35	0.06	35	3
Morocco	11	30	225	7	0.04	26	87
North Africa	15	60	105	6	0.05	33	300
Middle East	6	9	129	4	0.05	29	122
Avg of 91% of PR reserves	11	25	188	10	0.05	29	88
mg/kg of P	71	165	1,226	66	0.29	189	578
g/ha/yr applied with 20 kg P/ha	1	3.3	25	1	0.01	4	12

Table V.3.2. Parameters in wastes that should be analysed prior to assessing an application for a Paragraph 7 exemption. Source: SEPA 2015; The Waste Management Licensing (Scotland) Regulations 2011.

Waste category as in European Waste Catalogue (EWC)	Microbiology	PTE	Prescribed Substances
Food waste from food and drink manufacturers	✓	✓	✓
Wastes from agriculture, horticulture, aquaculture, forestry, hunting and fishing	✓	✓	✓
Wastes from the preparation and processing of meat, fish and other foods of animal origin			
-food unsuitable for consumption or processing	✓		
Wastes from pulp, paper and cardboard production			
-waste bark and wood	✓	✓	✓
-lime mud waste	✓	✓	✓
-sludges from on-site effluent treatment		✓	✓
-wastes not otherwise specified		✓	✓
Wastes from leather, fur and textile industry		✓	✓
Wastes from Manufacture, formulation, supply, and use (MFSU)of basic organic chemicals			
-sludges from on-site effluent not containing hazardous substances	✓	✓	✓
Wastes from thermal processes: wastes from power stations and other combustion plants		✓	✓
Construction and demolition wastes (including excavated soil from contaminated sites)			
-dredging spoil not containing hazardous substances		✓	✓
-soil and stones not containing hazardous substances	✓	✓	✓
Waste from wastewater facilities-off-site WwTP			
-waste from aerobic treatment -off-specification compost	✓	✓	✓
-waste from anaerobic treatment		✓	✓
Waste from the preparation of water intended for human consumption			
-sludges form water clarification	✓	✓	✓
Garden and park wastes (including cemetery waste)			
-biodegradable waste	✓	✓	✓
-soil and stones	✓	✓	✓

V.3.3. General and trace metal concentration ranges in composts, anaerobic digestate, biosolids and ash, struvite and biochar.

Tables V.3.3a-d summarise the general and trace metal concentration ranges in composts, anaerobic digestate, biosolids and ash, struvite and biochar. Then Table V.3.3 examines some scenarios (low and high representative contaminant ranges) for trace organic pollutants.

Table V.3.3a. Compost compositional ranges for dry matter (DM-% of Fresh Matter), organic matter (OM %DM), N & P concentrations (% DM) and PTE (mg/kg DM) of composts and ABP used as organic P fertilisers. Source: Möller et al.,2018

Type of compost	DM	OM	N	P	Cd	Cu	Ni	Pb	Zn	Hg	Cr
Green waste	52-74	23-51	0.7-1.6	0.14-0.32	0.19-0.7	106-213	12-36	17-50.7	22.3-50	5.7-23.5	0.05-0.16
Household waste	52-78	26-54	0.9-2.0	0.18-0.44	0.2-0.5	114-184	13.4-30.5	18.2-38	26.8-56.6	6.29-18.9	0.05-0.16
Household waste including catering and retailer organic wastes	52-77	26-52	0.9-2.0	0.18-0.44	0.2-0.7	114-280	13.4-42.2	18.2-67	26.8-80.9	6.29-27.8	0.05-0.17
Bone meal	92.5-97.3	16-53	1-7.7	6-16.5	0.2-0.3	0.5-50	1-24	0.4-10	89-133	0.04	0.3-37.5
Meat meal	92.5-95	54.4-98.9	5.8-15	0.3-4.7	0.4	0.05-29.4	0.8-3.1	4.3	47-140	0.2	5.9-8.3
Meat and Bone meal	91.2-99.3	47.1-91.1	3-12	2.2-9.6	0-1.7	0.19-26.5	0-38	0.01-36.2	28-174	0	1-9.7

Table V.3.3b. Anaerobic digestate ranges of values for the basic characteristics, nutrient contents and heavy metals. OFMSW: organic fraction of municipal solid waste. Source: Peng and Pivato 2019.

	OFMSW			Food waste		
	Whole	Solid	Liquid	Whole	Solid	Liquid
Basic characteristics						
pH	8.30	8.80	8.34–8.80	7.60–8.30	7.97	7.9
TS (%)	0.72–51.2	7.23–27.0	3.90	1.99–7.88	9.00	3.23
VS (%TS)	62.1	68.0–71.0	66.4	61.7–73.6	80.4	68.4
TC (%TS)	34.6	12.8–22.7	36.9	32.8–39.5	–	–
Nutrients contents						
$N–NH_3$ (g/L)	1.7–27.5	–	3.84	3.37	5.7	2.7
TKN (%TS)	2.79–14	–	–	15.4	–	–
C/N	1.3–29.8	12.1–20.9	2.7	2.63	–	–
TN (%TS)	1.3–12.4	1.09	13.85	4.15	3.97	8.65
P_2O_5 (%TS)	0.2–0.9	1.49	1.22	0.93	0.9	2.10
K_2O (%TS)	0.6–1.0	0.78	3.64	2.33	2.33	2.53
Heavy metals						
Cd (mg/kg)	0	–	–	<0.4	–	–
Pb (mg/kg)	15	–	–	9.8–36	–	–
Cu (mg/kg)	55	–	–	14–80	–	–
Hg (mg/kg)	–	–	–	<0.23	–	–
Ni (mg/kg)	50	–	–	11–20	–	–
Zn (mg/kg)	78	–	–	56–300	–	–
Cr (mg/kg)	188	–	–	6–40	–	–

Table V.3.3c. Biosolids range in values for DM (% of Fresh Matter), OM, P and N (% DM) and PTE (mg/kg DM). Source: Möller et al.,2018.

Biosolids	DM	OM	N	P	Cd	Cu	Ni	Pb	Zn	Hg	Cr
Liquid	1.7-6.5	42.7-78.1	4.7-17.2	2.4-5.6	0.1-1.3	85-679	15-72	8-99	310-1700	0.1-0.9	15-64
Dewatered	17.6-40.3	40.4-82.8	3.1-10.6	2.2-5.6	0.1-1.5	69-797	13-68	10-90	346-1400	0.1-0.9	16-257

Table V.3.3d. Struvite, Ashes and Biochar. DM (% of Fresh Matter), OM, P and N (% DM) and PTE (mg/kg DM). Source: Möller et al.,2018.

Parameter	Struvite	ABM ash (Untreated)	Sewage Sludge Ash (Untreated)	Sewage sludge ashes after Acid Digestion or leaching	Sewage Sludge Ashes after thermochemical treatment	Biochar
DM	55-57	97.8	-	-	-	
OM	-	-	-	-	-	
N	2.8-5.4	0.2	-	-	-	
P	6.1-13.5	6-19	6.7-9.4	5.7-11.2	7.7	5.3-5.9
Cd	0-1.76	0.3-1.3	0.3-3.5	0.1-0.2	0.3	
Cu	0.2-160	3.6-46.6	875-1240	10.5-249	601	420-952
Ni	0-28.6	2.1-78.4	64-68	32-72	56.4	46-77
Pb	0-44	8.7-17.6	58-141	0.1-6.7	60	30-100
Zn	1-403	16.3-373	31.7-2479	41-275	1710	530-2533
Hg	0-4.2	0	0.2	0	0.3	0.2-0.3
Cr	0-42	5.3-20.8	88-233	58-120	127	40-143

Table V.3.3e. Organic pollutants in mixed recyclable materials as dry matter masses for two contamination levels. Source: Weissengruber et al.,2018. PCB: polychlorinated biphenyls; PAH: Polycyclic aromatic hydrocarbons; PCDD/F: polychlorinated dibenzodioxins and polychlorinated dibenzofurans.

Contamination level	PCB (mg kg^{-1})		PAH (mg kg^{-1})		PCDD/F (ng TEQ kg^{-1})	
	Low	High	Low	High	Low	High
Green waste compost OF	0.01	0.04	0.38	6.40	0.003	0.005
Organic household waste compost OF	0.01	0.09	0.38	22.0	0.004	0.005
Biosolid	0.02	0.18	18.0	46.0	0.018	0.180
Organic household waste digestate OF	0.02	0.03	0.12	1.60	0.003	0.015
Meat and bone meal	n.a.	n.a.	n.a.	n.a.	0.0001	0.001

Appendix VI.1. Screening matrix for the catchment selection.

ID refers to the SEPA catchment location code; WFD failures for Soluble Reactive P (SRP) are denoted by M, Medium and P, Poor (whereas G, Good and H, High denotes attaining required thresholds); CAR sites are the number of licenced effluent discharges in the catchment; AD plants are the number of AD plants denoted on the WRAP website; Landcover denotes primary and secondary landcover classes from LCS88 (IG, Intensive Grassland; Ar, Arable; Rgrz, Rough grazing; Wood, Woodland; Moor, Moorland; Peat, Peatland; Urb, Urban). Sites are ordered in terms of the number of effluent sources. The selected four catchments are denoted in bold font.

			WFD failures for annual mean [SRP]												Landcover	
ID	**Name**	**Area km²**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**CAR sites**	**AD plants**	**Primary**	**Secondary**
6200	River Eden	61	M	M	M	G	G	G	M	M	M	M	5	0	IG	Rgrz
4500	River Devon	80	G	G	G	G	G	G	G	G	M	G	4	0	Ar	Urb
10610	River Nith	129	G	G	M	G	G	G	G	H	H	G	4	0	Ar	Wood
3100	River Avon	43	M	M	M	M	M	M	M	M	M	G	3	0	Ar	Rgrz
10427	**Lugar Water**	**54**	**M**	**M**	**M**	**M**	**M**	**M**	**P**	**M**	**M**	**M**	**3**	**0**	**Ar**	**Wood**
23241	Youlie Burn / Bronie Burn	71	M	M	M	M	M		M	M	M	M	3	0	IG	Moor
4000	**River Tyne**	**52**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**M**	**2**	**0**	**IG**	**Ar**
4001	River Tyne	55	P	P	P	P	P	P	P	P	P	P	2	0	IG	Wood
6201	River Eden	50	G	G	G	G	G	G	M	M	M	M	2	0	Ar	IG
6701	**River Farg**	**48**	**G**	**G**	**G**	**H**	**G**	**G**	**M**	**M**	**M**	**M**	**2**	**1**	**Ar**	**Wood**
23161	Idoch Water	94	M	M	M	M	M	M	G	G	G	G	2	1	IG	Urb
23221	North Ugie Water	51	M	M	M	M	M	G	G	G	G	G	2	0	Rgrz	Peat
23232	River Ythan	53	M	M	M	G	G	G	G	G	G	G	2	0	Wood	Peat
3021	Breich Water/Darmead Linn	48	M	M	M	M	M	M	M	M	M	M	1	0	Ar	Urb
3912	West Peffer/Mill Burn	41	P	P	P	P	P	P	P	P	P	P	1	0	IG	Rgrz
4205	**Bonny Water/Red Burn**	**53**	**P**	**P**	**P**	**M**	**M**	**M**	**P**	**P**	**P**	**M**	**1**	**1**	**Ar**	**IG**
5312	Lyne Water	40	H	H	H	H	H	H	M	M	M	M	1	0	Ar	Wood
5904	Vinny Water	53	M	M	M	M	G	G	M	M	M	M	1	0	Ar	Rgrz
10643	Pennyland Burn/Mein Water	40	G	G	G	G	G	G	M	M	M	M	1	0	IG	Ar
23061	Burn of Savoch	42	M	M	M	M	M	G	G	H	H	H	1	0	Urb	IG
23194	River Bogie	77	M	M	M	M	M	G	G	G	G	G	1	0	IG	Urb

			WFD failures for annual mean [SRP]												Landcover	
ID	Name	Area km²	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	CAR sites	AD plants	Primary	Secondary
23222	North Ugie Water	59	M	M	M	M	M	M	G	G	M	M	1	0	Rgrz	IG
23230	South Ugie Water	44	M	G	G	G	G	G	G	G	G	G	1	0	Rgrz	Wood
23234	Fordoun Burn	50	M	M	M	M	G	G	G	G	M	M	1	0	Peat	Moor
23269	River Don	49	M	M	G	G	G		G	G	H	H	1	0	Ar	IG
23284	Lochter Burn / Kings Burn	49	G	G	G	G	G	G	M	M	M	M	1	0	Ar	IG
23333	Beltie Burn	74	M	M	M	M	M	G	G	G	G	G	1	0	Ar	Wood
10000	White Cart Water	46	P	P	P	P	M	G	M	M	M	M	0	0	Ar	Wood
10042	River Clyde	216	H	G	G	G	H	H	H	H	M	M	0	0	IG	Wood
10052	Rotten Calder Water	56	G	G	G	M	G	G	G	M	M	M	0	0	Ar	Urb
10080	Nethan Water	59	G	G	G	G	G	G	M	M	M	M	0	0	IG	Urb
10394	Annick Water	65	M	M	G	G	G	G	M	M	M	M	0	0	IG	Moor
10493	Cross Water of Luce	62	H	G	G	M	G	G	H	H	H	H	0	0	IG	Rgrz
10739	River Annan	47	M	M	M	M	M	M	M	M	M	M	0	0	Ar	Peat
10928	Moffat Water	75		H	H	H	H	H	H	H	M	M	0	0	Ar	Wood
20038	Achairn Burn	42	G	G	G	G	G	G	M	M	M	G	0	0	Ar	IG
20306	River Nairn	52	H	H	H	M	M	M	M	M	M	H	0	0	Ar	IG
20309	Cawdor Burn	44	H	H	H	M	M	M	M	M	M	H	0	0	Ar	IG
20650	Sleach Water	60	M		M	M	G		M	M	G	G	0	0	IG	Wood
23075	Dullan Water	42	G	G	G	G	G	G	M	M	G	H	0	0	IG	Rgrz
23170	Keithny Burn / Forgue Burn	88	M	M	M	G	G	G	G	G	G	G	0	0	IG	Wood
23233	River Ythan	98	G	G	G	G	G	G	M	M	M	M	0	0	Moor	Peat
23239	Burn of Keithfield/ Raxton Burn	50	M	M	G	G	G	G	G	G	G	G	0	0	Peat	Wood
23310	Ton Burn / Cluny Burn lower	43	G	M	G	G	G	H	H	H	H	H	0	0	Ar	Wood
23320	Gormack Burn	59	M	M	M	H	H	H	G	G	H	H	0	0	Ar	IG
23338	Tarland Burn	74	M	G	G	G	G	G	H	H	H	H	0	0	IG	Wood

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