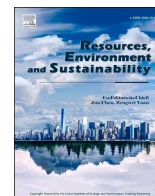




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Resources, Environment and Sustainability

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Research article

Phosphorus mobilization induced by bio-based fertilizers from soils with varying phosphorus retention capacities

Kari Ylivainio^{a,*}, Risto Uusitalo^a, Johanna Nikama^a, Andrea Bauerle^b, Antonio Delgado^c^a Natural Resources Institute Finland, Tietotie 4, FI-31600, Jokioinen, Finland^b Biobased Resources in the Bioeconomy, Institute of Crop Science, University of Hohenheim, 70599, Stuttgart, Germany^c Departamento de Agronomía, ETSIA, Universidad de Sevilla, Ctra Utrera km 1, 41013, Sevilla, Spain

ARTICLE INFO

Keywords:

Phosphorus mobility
Recycled fertilizers
Precipitates
Ash fertilizers
Soluble phosphorus
Turbidity

ABSTRACT

Phosphate fertilizer production relies on finite mineral resources. However, much of the phosphorus (P) taken up by crops eventually ends up in different waste streams. Ensuring P supply for agricultural sustainability requires a circular economy approach to producing bio-based fertilizers (BBFs) from these nutrient-rich side streams. This necessitates addressing not only the fertilizer value of BBFs but also the often-overlooked environmental impacts, which are surely affected not only by BBFs composition but also by soil properties. This study aimed to assess P losses promoted by BBFs, depending on their nature and the properties of soil to which they are applied, using a rain simulation after a 3-week incubation. To this end, 11 BBFs and triple superphosphate (TSP) were applied on soils originating from Finland (FI, pH 5.7), Germany (DE, pH 7.4), and Spain (ES, pH 8.1). The BBFs encompass a wide range of raw materials (plants, manure, animal by-products, sewage sludge) and production technologies (anaerobic digestion and composting, fermentation and distillation, various thermal treatments, and precipitation). The concentrations of dissolved reactive P (DRP) and molybdate-unreactive P (MU-P) in percolation water varied greatly depending on the soil properties and the P sources. Soil P tests that extract easily soluble P fractions predicted DRP concentrations in percolation water better than those extracting less soluble P fractions. The highest P concentration, especially DRP, was measured in the low P retention DE soil mixed with TSP. In acidic and calcareous soils, TSP, digested and composted pig slurry, and mineral/precipitate BBFs led to the greatest DRP mobilization. Organic BBFs containing tricalcium/hydroxyapatite and ash-based ones led to the least P mobilization in all soils. Granulation of BBFs effectively reduced P losses among BBFs with similar P speciation, while also offering improved handling and P utilization. Thus, compared with soluble mineral P fertilizers, BBFs offer a means to reduce P losses; however, minimizing P losses requires knowledge of both soil and BBF properties.

1. Introduction

The global population is projected to reach 9.7 billion by 2050, driving the need for increased food production and consequently raising the demand for fertilizer, including phosphorus (P), particularly in sub-Saharan Africa and Asia (Cordell and Neset, 2014; Helin and Weikard, 2019). Given that P is a finite and strategic resource, this calls for its sustainable use in agriculture, based on a circular economy approach and applying it only to fields that respond to P fertilization (Recena et al., 2022; McDowell et al., 2024). Furthermore, costs associated with the utilization of phosphate rock containing impurities such as Cd, fluctuating energy costs, and geopolitical issues are additional

arguments for the shift from linear to circular nutrient flows (Quitsov et al., 2025) and boost the utilization of nutrient-rich side streams (NRSSs), such as animal by-products and sewage sludges, as bio-based fertilizers (BBFs), to replace mineral P fertilizers (Hukari et al., 2016; Stamm et al., 2022). Replacing mineral fertilizers with BBFs is one way to extend P availability for food production (Barquet et al., 2020).

Overfertilization has rendered over 70 % of European croplands nonresponsive to P (Recena et al., 2022). Currently, NRSSs could almost meet Europe's P demand if applied only to fields where yield responses are expected, while simultaneously reducing environmental risks associated with nutrient accumulation in agricultural soils (Recena et al., 2022; van Dijk et al., 2016). However, the use of NRSS has challenges.

* Corresponding author.

E-mail addresses: kari.ylivainio@luke.fi (K. Ylivainio), risto.uusitalo@luke.fi (R. Uusitalo), johanna.nikama@luke.fi (J. Nikama), a.bauerle@uni-hohenheim.de (A. Bauerle), adelgado@us.es (A. Delgado).<https://doi.org/10.1016/j.resenv.2026.100297>

Received 28 August 2025; Received in revised form 12 January 2026; Accepted 12 January 2026

Available online 13 January 2026

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Products that can be used without processing, such as animal manures, with high water and low nutrient content, lead to high transportation costs (Chojnacka et al., 2020). Sewage sludge and industrial wastes often contain contaminants, requiring technological solutions to produce safe BBFs (Bünemann et al., 2024; Rigby et al., 2021). These technologies already exist and are at different stages of development (Ohtake and Okano, 2015), and BBFs with low organic contaminants content are now available on the market (Estoppey et al., 2024).

From an agronomic perspective, nutrients in fertilizers must be available to crops. Recent studies have shown that many novel P-BBFs are as effective as mineral fertilizers (Frick et al., 2025; Hernandez-Mora et al., 2024), partly due to the ability of crops to enhance P uptake through modified root morphology and the release of exudates (Richardson et al., 2009). However, the efficiency of the P source can vary drastically, for example, due to their variable (Fe + Al)/P ratios (Kiani and Ylivainio, 2024; Ylivainio et al., 2021). Furthermore, soil pH significantly affects not only fertilizer solubility and reactions but also the ability of crops to utilize P from different sources (Cabeza et al., 2011; Barrow, 2017; Degryse et al., 2017; Ylivainio et al., 2008).

Although simple laboratory tests can screen P solubility in fertilizers, further soil testing is necessary to assess the consequences of chemical reactions occurring in soils (Horta, 2015). Given the variability in BBFs and European soils, P solubility after application is likely nonuniform. The EU Fertilising Products Regulation (EU, 2019/1009) facilitates the use of CE-marked BBFs across the EU. However, understanding their impact on nutrient losses to surface waters is crucial for assessing their sustainable use by minimizing potential environmental impacts, especially when applied to soils with diverse characteristics. However, information on the P loss potential resulting from the application of BBFs of different types is scarce, which limits their sustainable use. Although they sometimes demonstrate good efficiency as P fertilizers (Frick et al., 2025), the main P forms present in BBFs are either organic or not readily soluble inorganic (Hernandez-Mora et al., 2024). Thus, these products can act as slow-release P fertilizers, and it can be hypothesized that they can reduce the risk of P loss, especially dissolved forms, compared with soluble mineral fertilizers.

The risk of P losses increases when soil P concentration exceeds the threshold values for crop response to fertilizer application, as determined by agronomic soil P tests (Withers et al., 2019). Thus, P loss risk is often assessed by relating Olsen-P to dissolved P concentrations (Bai et al., 2013; Casson et al., 2006; Heckrath et al., 1995; Pizzeghello et al., 2016). However, although the threshold value for a given soil P test aims to balance yield support without excessive P losses, soil properties such as texture, pH, and organic matter content affect the vulnerability to P losses (Bai et al., 2013; Pizzeghello et al., 2016). On the other hand, for soils that are similar in terms of P retention, different BBFs properties can lead to different P loss risk. Thus, assessing the impact of soil properties on P loss when different BBFs are applied is critical for practical recommendations on the sustainable use of these products.

The Fertilising Products Regulation (EU, 2019/1009) allows the use of CE-marked BBFs across the EU, whereas previously their use was restricted within individual member states. This normative change requires knowledge not only on their efficiency as fertilizers, but also on the effects of BBFs on P losses under various soil conditions to ensure their environmentally safe use. This study aimed to assess how BBFs promote losses of different P forms depending on BBF type (organic vs. inorganic), physical form (powder vs. granular), and the properties of the soils (acidic, neutral, and calcareous) to which they are applied. To this end, 11 different BBFs and three contrasting soils from Finland, Germany, and Spain were included, representing typical agricultural soils in these regions that receive P fertilization recommendations. The selection of these soils provides insight into the risk of P loss from BBF use across Europe by incorporating representative soils with different dominant P sorption processes and varying P retention capacities. To our knowledge, this is the first study at this scale to evaluate the combined influence of soil and BBF properties on P mobilization from soils. We

hypothesized that, owing to the heterogeneous compositions of BBFs, they may affect the solubility and mobilization of P in soils differently than conventional mineral fertilizers, and that these variable effects depend on the properties of both soils and the BBFs.

2. Material and methods

2.1. Fertilizer products and their analyses

This study included 11 BBFs from different NRSSs and production technologies (Table 1). The studied BBFs were either inorganic or organic fertilizers, most of which are already commercially available. The pyrolysis temperature for the pyrolyzed chicken manure (MBC) was 400–500 °C, and the composting period of digested, separated, and composted pig slurry (CPS) lasted about three months. A common inorganic P fertilizer, triple superphosphate (TSP), was included as a reference P fertilizer (Table 1).

For elemental analysis, the products were air-dried and milled to pass a 1 mm sieve. Phosphorus fractions of varying solubility in BBFs and TSP were determined using a modified Hedley fractionation scheme (Sharpley and Moyer, 2000), providing information on the relative distribution of P fractions in the products. Fractionation consisted of four sequential extractions: two with water, followed by 0.5 M NaHCO₃ (pH 8.5), 0.1 M NaOH, and 1 M HCl, using a 1:60 (w/v) ratio. Shaking was performed in an end-over-end shaker (27 rpm) for 16 h, except 4 h for the first water extraction. The samples were centrifuged (3000×g) for 15 min. Inorganic P (P_i) was analyzed from filtered (0.2 µm Whatman Nuclepore) supernatants, while total P was determined after autoclave digestion. Organic P (P_o) was calculated as the difference between total and P_i. Total P, Ca, Fe, and Al in the fertilizers were determined by using ICP-OES (PerkinElmer Optima 8300) after aqua regia digestion (concentrated nitric and hydrochloric acids) in a microwave oven (SFS-EN 13650). Nitrogen (N) and carbon (C) contents were determined using a LECO628 analyzer, and pH was measured in 1:5 (v/v) fertilizer-water suspensions.

2.2. Soils and analyses

Three different soils were used in the study: clay soils from Finland (FI; 60.82 N, 23.49 E) and Spain (ES; 37.41 N, 5.58 W), and a clay loam from Germany (DE; 48.71 N, 9.21 E) (Table 2). Soils represent common soil types found in these regions. All soils were taken from the plough layer (0–20/30 cm) of agricultural fields with soil P test values such that P fertilization is recommended according to respective national agronomic interpretations in each of these regions. The soils were air-dried at room temperature, passed through a 6-mm sieve, and then stored at +5 °C until the start of the experiment.

Subsamples were passed through a 2-mm sieve for soil texture analysis using the pipette method (Elonen, 1971) and for soil pH in a 1:2.5 (v/v) soil/water suspension. Before texture analyses, carbonates were removed with 10 % acetic acid. The carbonate content was determined using a Bernard calcimeter (Black, 1965). Soil C concentrations were determined using the LECO628 instrument and organic C concentration was calculated as the difference between the total and inorganic (CaCO₃) C.

The solubility of P in soils was analyzed using the following methods: the Olsen bicarbonate method (Olsen et al., 1954), calcium ammonium lactate (P-CAL; OENORM L 1087:2019–08-01, <https://www.dinmedia.de/en/standard/oenorm-l-1087/311897739>), acid ammonium acetate (pH 4.65) (P-Ac; Vuorinen and Mäkitie, 1955), the diffusive gradient in thin films (DGT; Santner et al., 2010), and electro-ultrafiltration (EUF; Németh, 1980). The concentrations of oxalate-extractable Al and Fe were determined with ICP-OES after extraction according to Schwertmann (1964). The P sorption index was calculated as the molar ratio of Olsen-P to oxalate-extractable Al and Fe. The total elemental analyses of P, Ca, Al, and Fe followed the same procedure as those used for

Table 1Phosphorus sources used in the rain simulation and their total elemental concentrations (g kg^{-1}) on a dry matter (DM) basis.

Acronym	Fertilizer	Physical form	PFC ^a	DM, %	pH	P	N	C	Ca	Fe	Al
BA1	Wheat and maize fermented and distilled	Granular	Organic	90.4	4.9	13.6	62.1	479	1.3	0.2	0.04
CPS	Digested, separated, and composted pig slurry	Compost	Organic	23.8	7.5	13.2	19.3	456	18.0	1.2	0.3
OPU	Chicken manure and grape residues	Granular	Organic	84.9	7.2	19.6	31.5	372	80.3	1.4	0.8
MB1	Meat and bone meal	Powder	Organic	97.6	6.2	63.4	81.3	362	129	0.2	0.04
MO14	Vegetable by-products, animal proteins, rock phosphate	Granular	Organic	91.7	6.5	67.4	24.9	238	166	2.0	1.5
MBC	Pyrolyzed chicken manure	Granular	Organic	98.3	10.5	30.6	29.4	363	181	1.2	0.6
EPH	Sunflower husk ash	Granular	Inorganic	97.1	10.8	19.5	1.3	57.7	105	1.8	1.5
PLA	Poultry litter ash	Powder	Inorganic	92.1	11.9	52.3	0.1	6.1	158	4.9	11.4
ADC	Sewage sludge treated with Na-AshDEC method	Powder	Inorganic	100	10.7	80.5	0.5	1.0	98.3	91.9	59.1
CGO	Struvite produced from sewage sludge	Granular	Inorganic	54.8	8.0	228	101	4.0	4.0	5.6	0.05
DCP	Dicalcium phosphate from sewage sludge ash	Powder	Inorganic	96.6	6.4	222	0.4	0.6	292	1.0	1.5
TSP	Triple superphosphate	Granular	Inorganic	93.0	2.8	221	0.5	6.5	191	1.3	1.9

^a Product function categories according to EU 2019/1009.**Table 2**

Soil properties of the rain simulation soils originating from Finland (FI), Germany (DE), and Spain (ES).

	FI	DE	ES
Texture, %			
Clay (<0.002 mm)	59.4	31.2	71.1
Silt (0.002–0.02 mm)	14.3	34.6	18.8
Fine sand (0.02–0.2 mm)	18.9	33.4	9.6
Sand (0.2–2 mm)	7.5	0.8	0.6
Organic C, %	4.5	1.4	0.2
CaCO ₃ , %	ND	ND	30.5
pH _{water} (1:2.5 v/v)	5.7	7.4	8.1
Electrical conductivity (EC), $\mu\text{S cm}^{-1}$	1.5	2.5	2.9
Soluble P concentrations			
Olsen-P, mg kg^{-1}	48.1	48.4	8.1
Acid ammonium acetate, mg l^{-1}	8.0	42.0	2.3
Calcium ammonium lactate, mg kg^{-1}	15	47	60
DGT ^a , $\mu\text{g l}^{-1}$	86	130	3
EUFB ^b , $\text{mg}/100\text{ g}$	2.0	4.6	1.1
Oxalate extractable elements, g kg^{-1}			
P	1.4	0.5	0.5
Fe	22.0	3.1	1.3
Al	4.1	1.5	1.9
Total concentrations, g kg^{-1}			
P	2.0	1.0	0.6
Ca	5.7	6.4	95.2
Fe	76.9	32.8	32.7
Al	59.7	40.4	55.7
Phosphorus sorption index, mmol kg^{-1}	0.003	0.014	0.017

^a Diffusive gradients in thin films.^b Electro-ultrafiltration.

fertilizers. The soil water-holding capacity (pF 2) was determined using pF plates.

2.3. Incubation

Each fertilizer product was mixed with 1.1 kg of soil (dry weight) at a total P rate of 100 mg kg^{-1} . A control treatment without any P application was included. All treatments were replicated three times. The soils were moistened with deionized water to 70 % of water holding capacity (pF 2) and kept at $20\text{ }^\circ\text{C}$ for three weeks under perforated plastic. The soil moisture was checked by weighing twice a week, and the evaporated water was replenished with deionized water. After the 3-week incubation period, 100 g of soil (dry weight) was removed and air-dried, followed by analyses of pH and P fractions according to the modified Hedley scheme described above, except that the soil samples for Hedley fractionation were ground with a ball mill.

2.4. Rainfall simulation and water analyses

For rainfall simulations, 1 kg of dry soil was placed in 15-cm PVC cylinders with a nylon mesh bottom. A 1-cm layer of coarse quartz sand

was added, followed by a 5–7 cm layer of incubated soil. The cylinders had drainage holes for collecting the percolated water. The setup was slightly compacted (bulk densities: FI soil 0.91 ± 0.03 , DE soil 1.21 ± 0.04 , and ES soil $0.90 \pm 0.04\text{ g cm}^{-3}$), moistened to pF 2, and left standing overnight loosely covered.

Rainfall was simulated using a dropformer-type rain maker (Uusitalo et al., 2012) at an intensity of 5 mm h^{-1} , representing a slightly higher average rainfall intensity but not a heavy rainfall in Europe (Li et al., 2020). Deionized water was pumped through 360 capillary tubes (0.51 mm i. d.) secured in a $1 \times 1\text{ m}$ steel frame to produce droplets with an average weight of 37.5 mg. The fall height was 2.4 m, resulting in a kinetic energy similar to that of natural rainfall due to the large droplet size compensating for lower impact velocity. The selected rainfall intensity did not cause flooding, and the soil was able to infiltrate the rain, as the main purpose was to collect percolated water through the soil column.

Percolated water was collected in acid-washed bottles, with three 100 ml samples taken from each unit. The water turbidity, pH, and electrical conductivity were measured immediately. The samples were frozen for later analysis. For dissolved reactive P (DRP), $\text{NO}_3\text{-N}$, and $\text{NH}_4\text{-N}$, the samples were filtered through $0.2\text{ }\mu\text{m}$ membranes before freezing. Total P and N concentrations were determined after autoclave digestion of unfiltered subsamples. Molybdate-unreactive P (MU-P) and organic N were calculated from the differences between total and dissolved forms. Thus, dissolved MU-P (also referred to as DOP, dissolved organic P) is included in the MU-P pool together with particulate P. For carbon measurements (total dissolved C, dissolved organic C (DOC), and inorganic C (IC)), samples were filtered through GF/C glass fiber filters.

The determination of $\text{PO}_4\text{-P}$, $\text{NO}_3\text{-N}$, and $\text{NH}_4\text{-N}$ were performed with a continuous flow analyzer (Lachat QuickChem 8500). Carbon concentration was determined using a Shimadzu TOC-V CSH Total organic C analyzer.

2.5. Statistical analyses

Data on soil (soil pH before rainfall simulation and Hedley fractionation) and percolation water quality obtained from rain simulations were analyzed using two-way ANOVA, with 0.05 used as the alpha threshold. The soils and fertilizers, and their interactions were used as factor for these analyses. The variables were log-transformed to normalize the residuals. Visual inspection was performed to check the correctness of the models (residual plots), equality of variances (homoscedasticity plots) and quantile distributions (Q-Q plots). Dunnett's post hoc test was used soil-wise to determine if the parameter values measured in the fertilized soil differed from those obtained from the untreated control soil or from the reference P source (TSP).

In the bar graphs, we used originally measured values and their standard deviations, but comparisons to control values (and TSPs) are plotted using transformed data with 95 % confidence intervals. GraphPad Prism 10.4.0 (GraphPad Software Inc., San Diego, CA, USA) was

used for data analysis.

3. Results

3.1. Soils

The five soil P tests gave variable results among the soils used in the study (Table 2). The FI and DE soils had equal and far higher Olsen-P values compared with the ES soil, whereas acid ammonium acetate extracted the highest P concentrations from the DE soil and CAL from the ES soil. The DGT and EUF methods gave similar ranks for soluble P concentrations, with the highest values for DE and the lowest values for ES soil. Thus, the ranking of soil P status clearly depended on the test used. The P sorption index was lowest in the FI soil, and it was about five-fold higher in DE soil.

The acidic FI soil had the highest organic matter content, while the calcareous ES soil had the lowest (Table 2). The oxalate extractable and total P, Fe, and Al concentrations were the highest in the FI soil, especially the Fe concentration (Table 2).

3.2. Soluble P concentrations of the fertilizers

The majority of P in fertilizers was found in an inorganic form; in 7 out of the 12 studied P sources the P_i share was greater than 98 % of the total sum of the P fractions (Fig. 1). The water-soluble P content differed widely between the BBFs, accounting for at least 50 % of the summed P fractions in only 2 products, whereas the share was 10 % or less in 8 products (Fig. 1). Only BA1 had a significant organic P content. Notably, in most of the BBFs, the least soluble fraction (HCl-P) made up the largest single fraction of extractable P (Fig. 1). The sum of extractable P by Hedley fractionation agreed well with the total P concentration by aqua regia digestion, with at least 72 % (93 % on average) of the total P being extractable.

3.3. Soil pH and P fractions in soils after incubation with BBFs

The effects of the BBFs on soil pH were most pronounced in FI, followed by DE and ES soils, where either 5, 3 or 2 products either decreased or increased the soil pH after 3 weeks of incubation in the corresponding soils (Table A1). Among the BBFs, BA1 decreased the soil pH to the lowest level in the DE and ES soils, whereas EPH increased it to

the highest level in all soils. The greatest variation in soil pH was observed in the DE soil, from 6.9 to 8.3.

According to Hedley fractionation, the water-soluble P_i concentration of unamended DE soil was far greater than that of the two other soils, but both 0.5 M NaHCO_3 - and 0.1 M NaOH-soluble P_i were greater in FI soil than in DE or ES soil. The ES soil showed more 1 M HCl-soluble P_i than the other soils (Table A1).

The majority of the P sources increased both the water- and bicarbonate-soluble P_i concentrations across all the soils. MO14 was the only BBF that did not enhance P_i solubility in either fraction in any of the soils; MB1 showed a similar pattern in the DE and ES soils. A significant increase in water- or bicarbonate-soluble P_o concentrations was observed only in the water-soluble fraction in the ES soil, where BA1 increased it to the highest level (Table A1).

The less soluble P fractions (0.1 M NaOH- and HCl-soluble P) accounted for the majority of the sum of the P fractions. In the FI soil, NaOH-soluble P, and in the DE and ES soils, HCl-soluble P_i constituted the highest share of the P fractions. Although the NaOH-soluble P_i concentration was variable among the unamended soils, from 10 mg kg^{-1} in the ES soil to 600 mg kg^{-1} in the FI soil, most of the P sources tended to increase this fraction. In FI soil, most applied P (100 mg kg^{-1}) was found in this fraction, whereas in the DE and ES soils, the respective fraction was HCl-soluble P_i . Among the total P added to the soils as MB1, more than 80 % was found in the HCl-soluble form in the DE and ES soils, whereas only about 10 % was present in this fraction in the acidic FI soil. In the FI soil, almost all P originating from MO14 was HCl-soluble. In the DE and ES soils, BA1 increased the NaOH-soluble P_o concentration to a significantly greater extent than the other P sources. Additionally, in the FI soil, BA1 tended to increase the NaOH-soluble P_o concentration.

3.4. Turbidity, electrical conductivity and pH of percolation waters

Each incubated soil unit was subjected to rainfall simulation, which yielded 300 ml of percolation water (the measured yield ranged from 274 to 319 ml, with a mean of 301 ml). Thus, the masses of the elements mobilized from the simulated units are directly comparable to the concentration differences in the percolation water. Although the total percolated water volume was similar among all the soils, the three soils differed in their susceptibility to start generating percolation water. Percolation through the FI, DE and ES soils started 1.7 ± 0.2 , 1.9 ± 0.2 ,

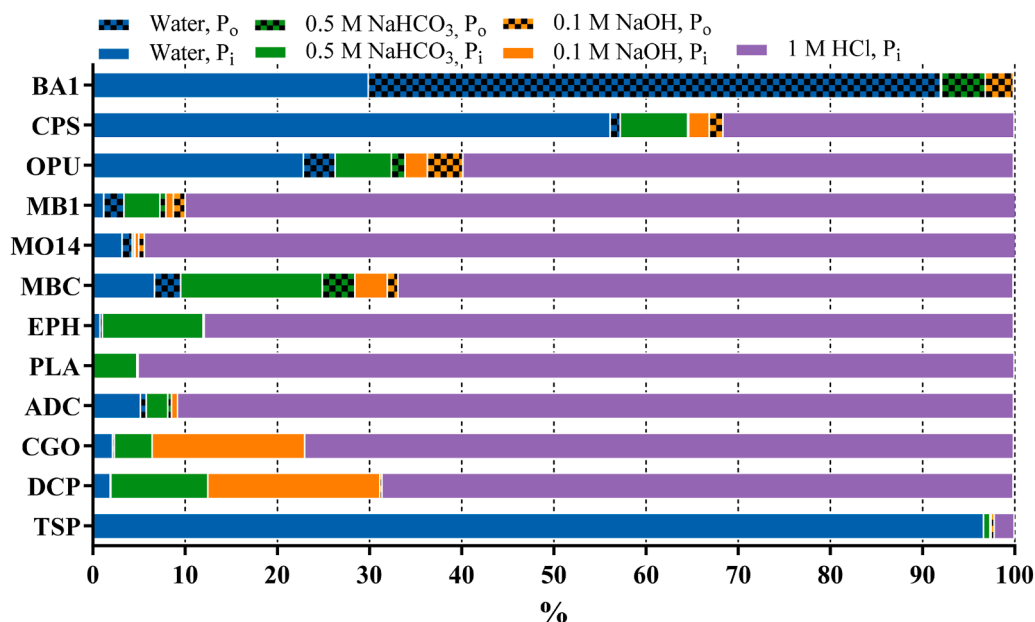


Fig. 1. Shares of P fractions in the studied P sources according to the Hedley fractionation scheme. See Table 1 for BBF abbreviations.

and 3.0 ± 0.3 h, respectively, after rainfall began. The FI and DE soils delivered the desired 300 ml of percolation water faster, 3.6 ± 0.5 and 3.4 ± 0.4 h, respectively, than the ES soil, which required 4.0 ± 0.9 h.

Among the three soils studied, the FI soil clearly produced the highest turbidity of water, followed by the DE and ES soils (Table A2), whereas electrical conductivity (EC) followed the opposite order ($ES > DE > FI$), and turbidity decreased with increasing EC (Fig. 2). For all the soils, a turbidity plateau was reached at $2000\text{--}3000 \mu\text{S cm}^{-1}$ in percolation water. Fertilizer products generally increased EC and decreased water turbidity. The effect on EC was modest when the fertilizer was mostly composed of minerals or precipitates (TSP, DCP, CGO, and ADC). Ash products (PLA and EPH) and organic P sources (BA1, CPS, OPU, and MB1) elevated the EC of percolation water the most, and BA1-amended soils showed the highest EC (Table A2).

The leachate pH reflected the original soil pH. However, BBFs increased the leachate pH in the acidic FI soil, whereas the effects were less pronounced in the DE and ES soils. The only exception to this was BA1, which substantially decreased the pH in both DE and ES soils (Table A2).

3.5. Phosphorus concentrations in the percolating waters

Among the three soils, the DE soil had the highest DRP concentration in the percolation water (Fig. 3, Table A2). In the nonfertilized controls, the DE soil had a mean DRP of 0.21 mg l^{-1} compared with 0.05 and 0.02 mg l^{-1} in the FI and ES soils, respectively. In all the soils, the greatest increases in DRP were measured for the TSP, DCP, and CPS, whereas BA1 also clearly increased the DRP concentration in percolation water from the DE and ES soils. The lowest DRP concentration in the percolating water, at the level of the control treatment, was measured for the soils incubated with the MB1, MO14, EPH, and PLA treatments.

The MU-P concentrations in the FI soil were generally higher than those in the DE and ES soils (Fig. 3, Table A2). Both ash products (EPH and PLA), MB1 and BA1 significantly decreased the MU-P concentration in percolation water collected from the FI soil, but only EPH decreased it when incubated in the DE and ES soils. Significant increases in MU-P concentrations were recorded only in the ES soil treated with MB1, BA1, and OPU. When all P sources were included, the correlation between turbidity and MU-P concentration was strongest in the FI soil and weakest in the ES soil. However, this correlation improved in both DE and ES soils when most soluble organic P sources (BA1, OPU, and CPS) were omitted from both DE and ES soils and less soluble MB1 from ES soil (Fig. A1). Because the dissolved MU-P pool was not determined separately, its actual size is unknown. However, soluble organic P-associations in BA1, OPU and CPS treatments may have inflated MU-P in the DE and ES soils, which produced percolates with much lower turbidity than the FI soil.

The total P concentration of percolation water was dominated by MU-P in all nonfertilized soils, most clearly so in the FI soil (the MU-P of TP was 93 % for FI, 60 % for DE, and 88 % for ES soil). The share of MU-P

was greater than that of DRP in the FI soil independent of the P source. For the DE soil, DRP was the major contributor to TP in 6 out of the 12 treatments; EPH, ADC, CGO, and CPS caused rather similar (55–61 %) DRP losses, whereas DCP (71 %) and TSP (82 %) caused the highest DRP share among the P sources (Table A2). In the ES soil, DRP was the major fraction of TP in the TSP, DCP, and CPS treatments. The lowest TP concentrations were observed with EPH for all soils. No significant increases in TP were observed for the FI soil, but four (TSP, DCP, BA1, and CPS) and seven P sources (TSP, DCP, ADC, MB1, BA1, OPU, and CPS) increased the TP concentrations in the water percolating through the DE and ES soils, respectively (Fig. 3, Table A2).

Data on N and C in percolation waters are shown in Supplemental Materials.

4. Discussion

4.1. Assessing P mobilization from soils with different P retention capacities

Soil P tests are found useful not only for assessing P fertilization needs but also for evaluating potential P loss risks (Withers et al., 2019). However, in our study, the Olsen-P and CAL methods, which extract less soluble P forms, did not accurately predict DRP concentrations in the percolation water. For example, in the control soils, Olsen-P (pH of the extractant 8.5) extracted similar amounts of P from the FI and DE soils, yet the DRP concentration in the percolating water was approximately fourfold greater in the DE soil. The acidic extractant CAL (pH of the extractant 4.1) had the highest value in the ES soil, but this soil produced the lowest DRP concentration in percolation water. This was expected because CAL dissolves insoluble Ca phosphates present in this soil. These P forms do not readily contribute to the DRP concentration in percolating water. In contrast, milder extractants such as acid ammonium acetate and water-soluble fractions from sequential P fractionation, better predicted DRP levels, as they primarily extract readily soluble P. Additionally, DGT and EUF belong to this same category of milder P tests. This ranking is supported by the linear relationship between soil P test values and DRP concentration in the control treatments (Fig. A5). Our results suggest that methods extracting readily soluble P fractions are best for predicting DRP loss risks, whereas less soluble P fractions can contribute to bioavailable P for crops and need to be considered when planning P fertilization.

The labile P fraction, which includes both the water- and the 0.5 M NaHCO_3 -extractable P fraction, represents the plant-available P concentration in soils. The latter extractant (0.5 M NaHCO_3) is the same as that in the Olsen-P method, but the wider soil-solution ratio and longer extraction time probably explain the higher extracted P concentration compared to the Olsen-P values in all soils. This labile P pool was greater in the FI than in the DE soil, indicating that the labile soil P concentration may not necessarily correlate with P mobilized in discharge waters from soils with different P retention capacities, but rather relates

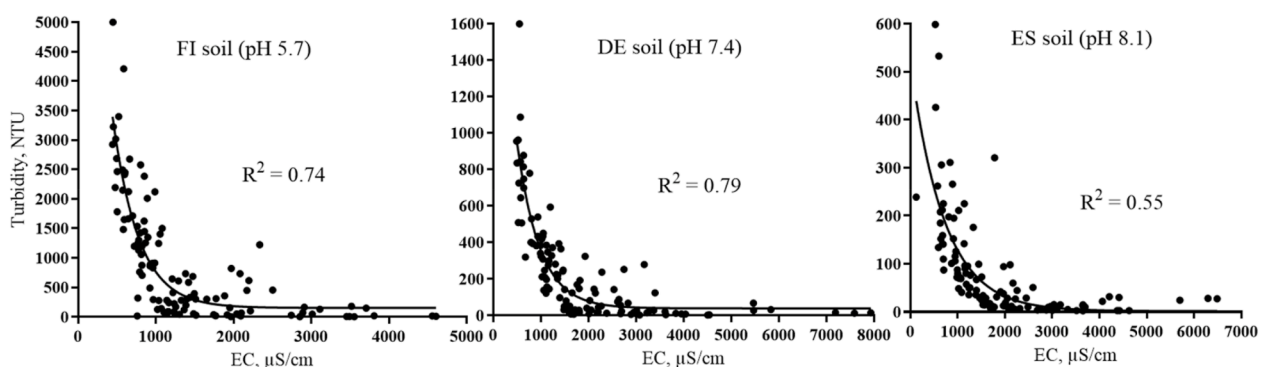


Fig. 2. Relationships between turbidity and electrical conductivity (EC) in percolation water from soils originating from Finland (FI), Germany (DE), and Spain (ES).

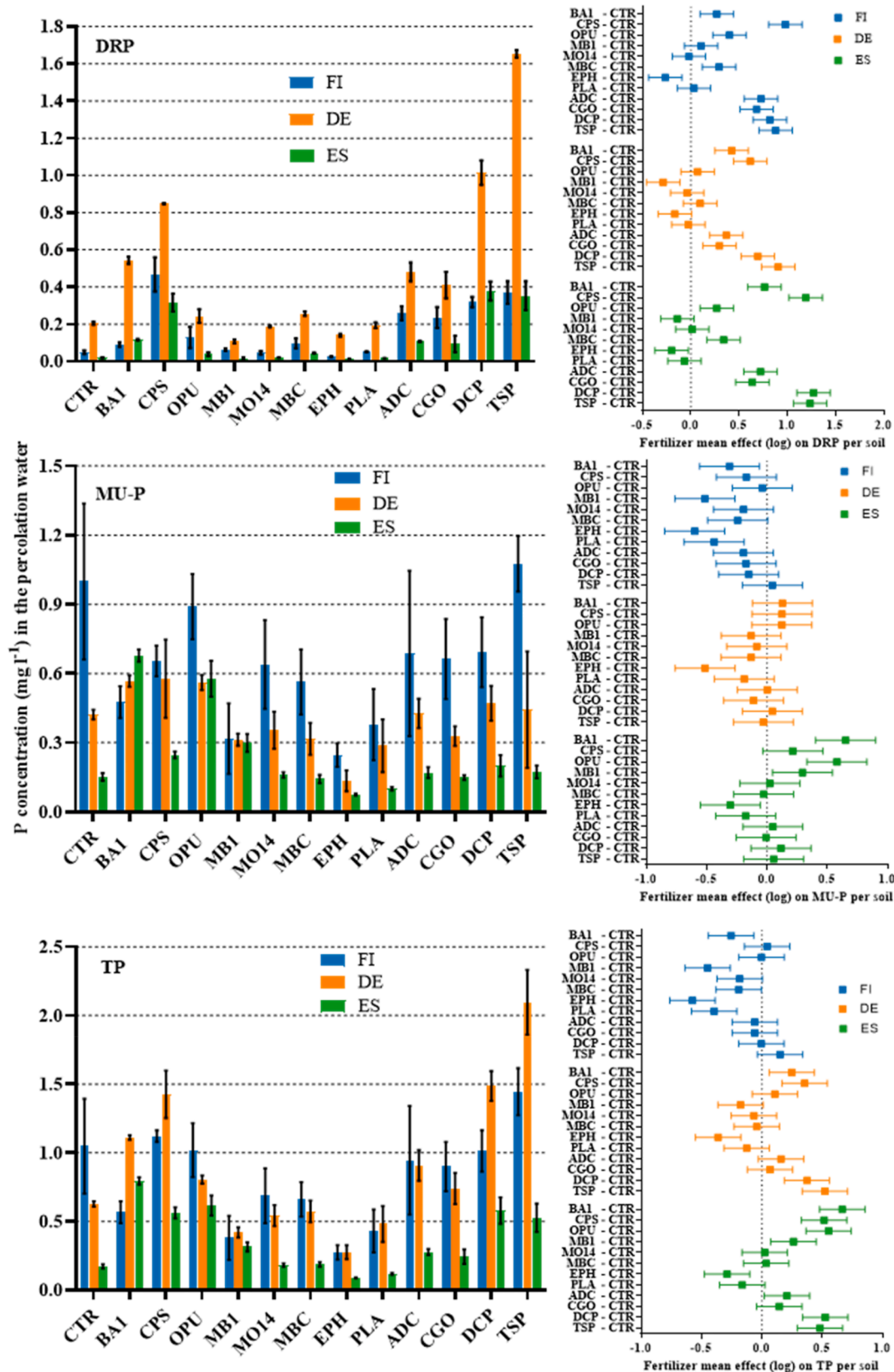


Fig. 3. Left-hand panels: dissolved reactive P (DRP), molybdate-unreactive P (MU-P), and total P (TP) concentrations in percolation waters collected during rainfall simulations (all have the same y-axis units); the graphs show original data with standard deviations ($n = 3$). Right-hand panels: differences in P concentrations in leachates between fertilized treatments and the control (nonfertilized soil); log-transformed data with 95 % confidence intervals (Dunnett test). The dotted vertical lines represent the means of the controls. See Table 1 for an explanation of the BBF abbreviations.

to the stock of plant-available P, as it was originally designed to do.

Soil characteristics affect the interpretation of soil P tests for assessing DRP loss risk, which may challenge Europe-wide regulations such as the Soil Monitoring Law (EU, 2025/2360). In this law, one of the soil descriptors for healthy soil is the extractable P concentration, intended to ensure that no harm to human health or the environment occurs. These maximum values for soil P status are likely determined using soil P test results, and, as demonstrated in this study, similar soil P test values can lead to widely different DRP losses depending on soil properties. German (DE) soil, with its low Fe and Al concentrations and absence of free CaCO₃, has a limited P retention capacity and is therefore prone to producing high DRP concentrations in percolating water, particularly from the easily soluble TSP. In the acidic FI soil, soluble P is adsorbed mainly onto the surfaces of Fe- and Al-oxides (Peltovuori, 2007) and is mostly extractable with 0.1 M NaOH-P (as in the FI soil of this study). Olsen-P extraction solution with a pH of 8.5 can solubilize some of this P pool by changing the soil surface charge to be more negative, enhancing P desorption (Hingston et al., 1967). In the calcareous ES soil, P is present mainly in the form of Ca phosphates or is adsorbed on carbonates (Frossard et al., 1995), and detected as a high share of acid (1 M HCl)-soluble P.

When comparing different soils, Olsen-P alone does not appear to be a reliable indicator of P loss risk because the relationship between Olsen-P and the readily desorbable P fraction varies depending on soil texture, mineralogy, and pH (Recena et al., 2022). Therefore, Olsen-P is sometimes coupled with a soil P sorption index that accounts for differences in soil P sorption capacity (Bache and Williams, 1971). In this study, the ratio of the molar concentration of Olsen-P to oxalate-extractable Al and Fe, used as an estimate of P sorption capacity, yielded approximately the same relationship between FI and DE soils as did P extraction with acid ammonium acetate. However, this sorption index is not suitable for calcareous soils, where Al- and Fe-oxides do not control P solubility.

4.2. Effects of inorganic P sources on DRP loss

In rain simulations, P mobilization varied significantly depending on the properties of both fertilizer and soil. Inorganic P sources, such as TSP and DCP, increased the most DRP mobilization in all soils, whereas the ash-based fertilizers (EPH, PLA) had only a small effect on DRP. Generally, intermediate effects were found for the organic fertilizers, except for CPS. In all treatments, DE soil showed particularly high DRP mobilization due to its low P adsorption capacity, aligning with studies showing that TSP causing greater DRP losses than manure-based sources in soils with low P retention (Kang et al., 2011). For soils with low P retention capacity, sparsely soluble P fertilizers have been recommended to reduce P leaching (Summers and Weaver, 2022).

DRP concentration in leachates from CGO (struvite)-amended soils remained lower than that of the TSP-amended soils, despite CGO being associated with the highest water-soluble P_i concentration in the incubated soils prior to rainfall simulation. Both CGO and TSP were added in granular form, but the solubilization of CGO was expected to be slower than that of TSP because CGO had far lower water-soluble P_i content *per se*, 2 % in CGO and 97 % in TSP. The high water-soluble P_i concentration in the soil incubated with CGO and the discrepancy between water-soluble P_i and DRP, were probably due to the soil samples being milled prior to Hedley fractionation, which increased struvite dissolution during P fractionation. This agrees with Degryse et al. (2017) who observed that struvite granules dissolved slowly in alkaline soil, whereas the powder form of struvite dissolved quickly. In addition, CGO caused lower DRP losses in the calcareous ES soil than in the FI and DE soils, supporting the view that the solubility of granular struvite decreases as soil pH increases (Degryse et al., 2017). Similar patterns were observed for the dicalcium P-containing fertilizers, DCP, ADC, and (the mainly organic) OPU (Hernandez-Mora et al., 2024), which differed widely in their DRP mobilization. Mineral DCP fertilizer, applied in powder form, mobilized the second most DRP (after TSP) from the DE soil, but clearly

less from the FI and ES soils. In turn, granular OPU caused relatively little DRP mobilization in all soils. This study clearly demonstrates that caution is required when interpreting potential DRP losses based solely on P solubility in milled soils, as leaching risk is influenced by the form of the applied P fertilizers. Hedley fractionation therefore provides only P solubility based on P speciation in a milled sample and does not account for the physical form of the BBF.

The ash-based products (EPH and PLA), for which DRP mobilization was not very different from that of the unamended control soils, mainly contained acid-soluble P and less than 12 % of P in labile forms. The most commonly used compliance testing method for mineral P fertilizers is neutral ammonium citrate (NAC), which, according to Hernandez-Mora et al. (2024), extracts 80.5 % and 35.2 % of the total P from EPH and PLA, respectively. Despite the lower NAC solubility, PLA tended to cause greater DRP mobilization than EPH from the FI and DE soils. Although both products are strongly alkaline, only EPH increased the soil pH. This is due to the 2.5-fold higher application rate of EPH than of PLA to achieve equal applied P rates. The high applications also resulted in higher EC in the percolating water, suggesting a bolstered salt effect on the electrical double layer in soil and thus reduced DRP mobilization (*cf.* Uusitalo et al., 2012).

4.3. Effects of organic P sources on DRP loss in different soil types

Among the organic BBFs, CPS treatment caused the greatest DRP losses, comparable to those of the TSP treatment in the FI and ES soils. This is likely due to its high water-soluble P_i content, accompanied by high C application rates that may cause competition for P adsorption sites (Haynes and Mokolobate, 2001). Unlike CPS, fermented plant residues (BA1) and pelletized poultry manure (OPU) contain easily degradable organic matter (Agostini et al., 2024), providing soluble P after mineralization. BA1 showed relatively high DRP mobilization in the DE soil, but less so in the FI and ES soils. This is possibly due to the high water-soluble P_o content, which mineralized into P_i during incubation, likely adding to DRP in percolates or competing for the same adsorption sites with P_i when mixed with a soil with low P sorption capacity. In turn, the high acid-soluble P_i share (60 %) of OPU did not translate to an increase in DRP, especially in the ES and DE soils with basic soil reaction.

Hydroxyapatite/tricalcium phosphate-containing MB1, MO14, and MBC caused similarly very moderate DRP mobilization in all soils. The water-soluble P_i concentration in soils, which has been shown to be a good indicator of P loss risk (Sharpley and Moyer, 2000), was greater in MBC-treated soils than in MB1 and MO14-amended soils prior to rain simulation. MBC also promoted some DRP mobilization. Compared with OPU, MB1, and MO14, the organic matter in MBC appeared to be more resistant to mineralization, as shown by the low DOC concentrations in the leachates, despite the relatively high total C input with MBC (Table A3 and Fig. A3). Organic products with high acid-soluble P content, such as MB1 and MO14, typically show limited DRP mobilization, but P solubilizes over time in acidic soils. Apatite-based P sources are generally less soluble in calcareous soils (Delgado et al., 2002) and markedly increase the acid-soluble P_i pool, as observed with MB1 in the ES soil. However, mixing MO14 in the DE and ES soils resulted in less accumulation in acid-soluble P_i, which suggests that it contained stable P_o or less soluble P_i forms when mixed with these soils. In contrast, incubation of MO14 in the FI soil markedly contributed to the acid-soluble P_i pool, corresponding to 96 % of the applied P.

4.4. Selection of BBFs for minimizing P losses from different soils

For total P mobilization, most BBFs reduced TP mobility in the FI soil, whereas an increase in TP was observed more frequently than decreases in the DE and ES soils. The different behaviour of the FI soils was due to the tendency for lower mobility as MU-P form in BBF-amended soil, especially with the organic amendments (BA1, CPS,

OPU, and MB1). In contrast, the opposite pattern was observed in the DE and especially in the ES soil, possibly because dissolved MU-P was included in the MU-P pool. The ash products, especially EPH, were the only BBFs that decreased both TP and MU-P mobilization in all soils.

Mobilization of MU-P in the FI soil was clearly associated with the EC of percolated water, and to some degree also in the DE and ES soils, depending on the amounts of soluble salts added with the fertilizers (see Uusitalo et al., 2012). Although there was a clear relationship between EC and turbidity in all soils, the correlation between turbidity and MU-P decreased in the following order: FI soil ($R^2 = 0.87$) > DE soil ($R^2 = 0.64$) > ES soil ($R^2 = 0.01$) when all the treatments were combined. In addition, P_o can significantly contribute to P losses (Ron Vaz et al., 1993), and in this work, P_o (encompassing dissolved MU-P) was included in the MU-P fraction in the percolation water analysis. The contribution of dissolved P_o forms is supported by the highest DOC and MU-P concentrations observed in the BA1 and OPU treatments. Then, turbidity alone may be a poor indicator of P loss in calcareous environments, as with equal turbidity, organic BBFs caused higher MU-P losses than inorganic BBFs (Fig. A1). High turbidity in the acidic FI soil is partly related to a low concentration of calcium, the major cation involved in enhancing clay aggregation. This was clearly observed when the rain simulation progressed and turbidity increased more in the FI soil than in the other soils (Fig. A2), suggesting the dispersion of high P adsorbing clay particles, leading to P losses as MU-P.

In the FI and DE soils, the overall effect of BBFs on MU-P was either decreased MU-P mobilization or no significant changes. However, for the ES soil, the organic fertilizers, especially BA1 and OPU, increased MU-P mobility the most. These fertilizers also ranked highest when considering their increasing effect on C and N mobilization (Table A2, Table A3, Fig. A3, and Fig. A4). We speculate that in the calcareous ES soil, these effects may be associated with higher overall mineralization (Curtin et al., 1998; Stewart and Tiessen, 1987), or with low organic P adsorption in the calcareous soil.

For the control treatment, most mobilized P was MU-P, and this fraction may initially seem the most important in determining the overall goodness of BBF-soil fits. However, the changes in DRP due to BBFs were often greater than those associated with MU-P, especially in the DE soil. Of the P forms, DRP is more bioavailable (Ekholm and Krogerus, 2003), and consequently more harmful in water environments, which emphasizes its importance in decreasing the eutrophication of surface waters. Ideally, BBFs that reduce both DRP and MU-P mobilization would be the best fit products. Although erosion is the primary pathway for phosphorus (P) losses under field conditions, particularly as MU-P following heavy rainfall, subsurface leaching as DRP can also contribute significantly to total P losses. In this study, the proportion of DRP losses relative to total P losses followed a trend similar to that observed in field studies (Díaz et al., 2013; Siebers et al., 2023; Turtola and Paajanen, 1995).

Overall, the ash products EPH and PLA were the optimal solutions for all soils. For the FI soil, the least preferred products would be the mineral/precipitate products (DCP, CGO, ADC, TSP) and (organic) CPS because of their increasing effect on DRP mobility and only small, if any, decrease in MU-P mobility. For the DE soil, the highest P mobility was measured for TSP, DCP, CPS, and BA1. They all clearly increased DRP, but either slightly increased MU-P mobility or did not affect it at all. For the ES soil, increases in both DRP and MU-P were measured for the organic BBFs (BA1 and OPU), whereas MO14 had no effect on either DRP or MU-P losses, likely due to P supplementation through apatite. Mineral/precipitate BBFs clearly increased DRP but did not affect MU-P mobilization.

Fertilization efficiencies of the studied BBFs were evaluated in pot trials (Hernandez-Mora et al., 2024). BBFs containing hydroxypapatite/tricalcium phosphate and phytic acid (EPH, PLA, MO14, MB1, MBC, BA1) as the main P species generally exhibited lower fertilization efficiencies than TSP. However, all tested BBFs, except CPS, DCP, and MBC, were included in a two-year field trial conducted across Europe

under varying climatic conditions (Frick et al., 2025). The agronomic efficiencies of these BBFs were comparable to TSP, supporting the view that BBFs can serve as environmentally friendly alternatives to mineral P fertilizers while maintaining crop productivity.

5. Conclusions

The soil P test methods extracting easily soluble P fractions (acid ammonium acetate, DGT, and EUF methods) predicted P mobilization better than the commonly used methods extracting less soluble P fractions (Olsen-P, CAL). This study clearly reveals that soil chemical properties must be considered when selecting soil P test methods for predicting P losses. This is especially necessary, as the new Fertilising Products Regulation (2019/1009) ensures the free movement of CE-marked fertilizers across the EU. Agricultural soils with low P retention capacity may be at risk of higher proportion of DRP losses out of total P losses. In these soils, potential risks for P losses should be estimated via soil P test methods that extract easily soluble P fractions or by analyzing the soil's P retention capacity.

Although the chemical properties of recycled fertilizers produced from NRSSs vary greatly, they cause lower P mobilization than does TSP in most cases. Phosphorus losses from BBF-treated soils were not dependent on P solubility in the products but were strongly affected by the soil properties and physical form of the fertilizer. From an environmental perspective, all the studied BBFs caused far lower DRP losses than easily soluble TSP in soils with low P retention capacity. This goal is best achieved using ash-based and organic BBFs with low P solubility in both acidic and calcareous soils with high P retention capacities. Composting and pyrolysis of nutrient-rich side streams may be beneficial technologies for increasing the organic matter content in calcareous soils and improving soil quality through carbon sequestration, whereas easily mineralized BBFs contributed significantly to P losses. Although the incubation period lasted only 3 weeks, a short time frame is likely to reflect the risk of P losses following fertilization. As time progresses, adsorption or precipitation converts applied P into less soluble forms, thereby reducing DRP losses, whereas less soluble BBFs may remain in insoluble forms depending on soil properties. Therefore, in the short term, BBF properties likely have a stronger impact on P losses, whereas in the long term, variation in P losses among different BBF types decreases. Granulation of BBFs effectively reduced P mobilization from BBFs compared with products with similar P speciation but in a powder form. Granulation also improves the handling of BBFs and makes them more suitable for placement directly beneath seeds during planting, thereby ensuring that P is immediately available to growing plants and reducing the risk of leaching.

CRedit authorship contribution statement

Kari Ylivainio: Writing – original draft, Visualization, Validation, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Risto Uusitalo:** Writing – review & editing, Visualization, Validation, Methodology, Formal analysis, Data curation. **Johanna Nikama:** Writing – review & editing, Methodology, Investigation. **Andrea Bauerle:** Writing – review & editing, Validation, Resources, Methodology, Investigation, Data curation. **Antonio Delgado:** Writing – review & editing, Validation, Resources, Methodology, Investigation, Data curation.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Kari Ylivainio reports financial support was provided by European Commission. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could

have appeared to influence the work reported in this paper.

Acknowledgments

This project received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 818309 (LEX4BIO). The results reported in this paper reflect only the authors' views, and the European Commission is not responsible for any use that may be made of the information it contains.

Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.resenv.2026.100297>.

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