nitrogen

MDPI

Article

Soil-Specific Nitrogen and Phosphorus Leaching Patterns of Recovered Nutrients

Bárbara Palacino *, Paula de la Sen and Trinidad De Marco

Fundación CIRCE—Centro de Investigación de Recursos y Consumos Energéticos, Parque Empresarial Dinamiza, Avda. Ranillas 3D, 1ª Planta, 50018 Zaragoza, Spain; pdelasen@fcirce.es (P.d.l.S.); tdemarco@fcirce.es (T.D.M.)
* Correspondence: bpalacino@fcirce.es

Abstract

Products obtained from livestock manure and anaerobic digestion are increasingly promoted as alternatives to conventional mineral fertilisers. However, their environmental performance depends on fertiliser characteristics and soil properties controlling nutrient mobility. This study evaluated nitrogen (N) and phosphorus (P) leaching dynamics following the application of pig slurry, digestate, and ammonium sulphate recovered through digestate stripping. Replicated soil column experiments were conducted using four agricultural soils from three European river basins (Ebro, Lielupe and Danube), three fertiliser treatments, and three replicates per treatment, under controlled climatic conditions (15–25 °C; 50–60% relative humidity; 16 h light/8 h dark cycle) reproducing field climate and water application regimes. Leachates were periodically collected and analysed for ammonium (NH_4^+-N), nitrate (NO_3^--N), and phosphate (PO_4^{3-}-P). Results showed distinct nutrient release and transport patterns among fertilisers and soils. Ammonium sulphate produced rapid NH_4^+-N losses immediately after application, whereas digestate and pig slurry exhibited a more gradual nutrient release. The relative contribution of ammonium and nitrate to total N leaching varied among soils, highlighting the influence of soil properties on nitrification and nutrient retention processes. Phosphorus losses remained low across treatments. These findings support nutrient recovery strategies adapted to soil conditions to enhance nutrient circularity while minimizing environmental losses. In practical terms, managers and policymakers should require soil-specific nutrient management plans for recovered fertilisers, prioritising slow-release products such as digestate or pig slurry on soils prone to nitrogen leaching, and restricting ammonium sulphate to soils with lower leaching risk.

Keywords: nutrient recovery; pig slurry; digestate; ammonium sulphate; nitrogen leaching; soil health

Check for updates

Academic Editor: Arbindra Timilsina

Received: 31 August 2026
Revised: 27 September 2026
Accepted: 1 October 2026
Published: 6 October 2026

Copyright: © 2026 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license.

1. Introduction

Soil degradation is one of the major environmental and agronomic challenges facing Europe. Currently, around 60% of European soils are considered unhealthy due to erosion, contamination, loss of organic matter, decline in soil biodiversity, and nutrient imbalances, compromising their functionality and the provision of essential ecosystem services that support agricultural production and resilience to climate change [1–3]. Nutrient mismanagement has been identified as one of the major threats to soil functionality worldwide, affecting nutrient cycling, water quality, crop productivity and environmental sustainability [4]. While nutrient depletion remains a constraint in some regions, excessive nutrient

accumulation and associated nutrient losses are increasingly recognised as a major challenge in intensive agricultural systems. Improving nutrient use efficiency while reducing environmental losses has become a central objective of sustainable soil management and soil health restoration strategies.

At the same time, Europe generates large quantities of organic residues and nutrient-rich effluents, including livestock manures, anaerobic digestion by-products, dairy industry residues and by-products, as well as sewage sludge and compost derived from wastewater treatment [5,6]. These streams contain significant amounts of nitrogen (N) and phosphorus (P), two essential nutrients for crop production whose management is critical from both agronomic and environmental perspectives. While adequate nutrient supply is necessary to sustain crop productivity, excessive nutrient application or inefficient nutrient use may result in nutrient losses to the environment. Nitrate pollution originating from agricultural activities remains one of the major pressures affecting groundwater quality in many agricultural regions, while phosphorus losses contribute significantly to eutrophication processes in surface waters [7]. These environmental concerns highlight the need for more efficient nutrient management strategies that maximise nutrient use efficiency while minimising losses to the environment.

The recovery and reuse of N and P represent an important opportunity to close nutrient cycles, reduce dependence on mineral fertilisers and improve resource efficiency within the framework of the European Circular Economy Strategy [8,9]. In this context, the use of organic fertilisers and recycled nutrient products has gained increasing attention as part of the transition towards circular agricultural systems and more sustainable nutrient management strategies [7–10]. Among the different nutrient recovery alternatives currently available, pig slurry and digestate are two of the most widely used organic fertilisers in European agriculture, particularly in regions characterised by intensive livestock production. In the EU-27 and UK alone, livestock farming generates more than 1.4 billion tonnes of manure annually, of which over 90% is directly reapplied to agricultural soils [11], while European anaerobic digestion plants produced an estimated 25 million tonnes of digestate (dry matter) in 2024, with the potential to replace up to 17% of the EU's nitrogen-based mineral fertiliser demand [12]. Pig slurry is traditionally applied as a source of both organic matter and nutrients, whereas digestate, generated during anaerobic digestion processes, has gained increasing relevance due to its dual role in renewable energy production and nutrient recycling. Additionally, recent nutrient recovery technologies such as ammonia stripping enable the recovery of nitrogen from digestate in the form of ammonium sulphate, a concentrated recycled fertiliser that may partially substitute conventional mineral N fertilisers [13,14]. Compared with pig slurry and digestate, ammonium sulphate contains nitrogen in a more plant-available form, resulting in nutrient release patterns that more closely resemble those of conventional inorganic fertilisers while maintaining the benefits associated with nutrient recovery and recycling [14]. Although all three products contribute to nutrient recycling, they differ considerably in nutrient concentration, nutrient speciation and nutrient release dynamics, potentially resulting in contrasting environmental behaviour after soil application.

The environmental fate of nutrients depends largely on their chemical form and the transformations they undergo after application to soil. Nitrogen can be present as ammonium (NH_4^+-N) or nitrate (NO_3^--N), whereas phosphorus is predominantly found as phosphate (PO_4^{3-}-P). These nutrient species differ largely in their mobility and retention mechanisms. Ammonium may be temporarily retained through cation exchange processes, whereas nitrate remains highly mobile in the soil solution and is more susceptible to leaching following nitrification, particularly under conditions favouring water percolation and drainage [15–19]. Phosphate exhibits lower mobility and its availability is controlled

by adsorption, desorption and precipitation reactions involving soil minerals and organic matter [16]. Subsequently, nutrient retention, transformation and transport processes determine the extent to which recovered nutrients remain available for crop uptake or are lost from the system. These processes constitute fundamental components of soil nutrient cycling, one of the key functions supporting soil health and ecosystem productivity. The main nutrient transformation, retention and loss pathways considered in this study are summarised in Figure 1.

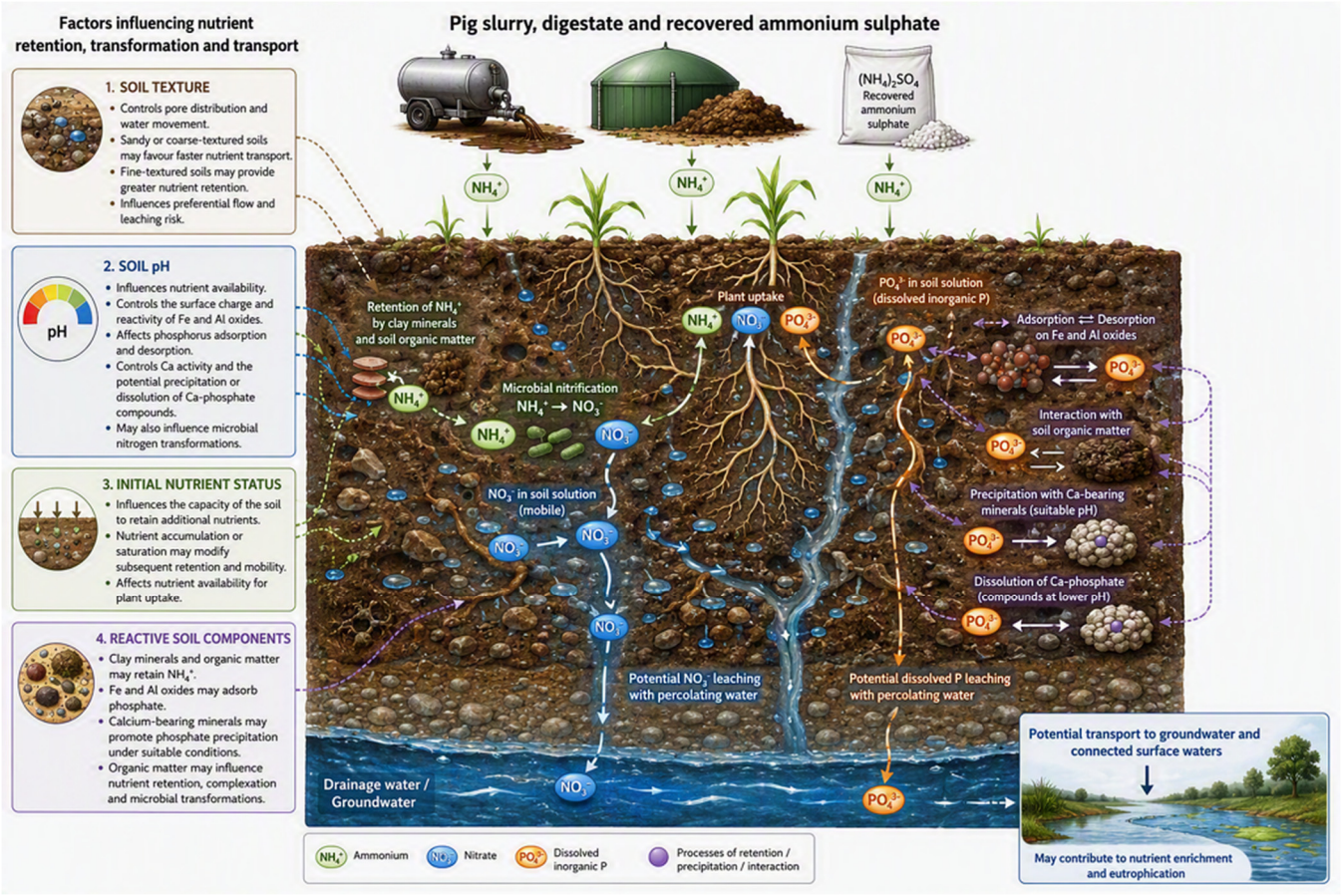


Figure 1. Conceptual framework of nitrogen and phosphorus dynamics in soils receiving pig slurry, digestate and recovered ammonium sulphate.

However, the agronomic performance and environmental behaviour of recovered nutrient products depend not only on fertiliser composition but also on the physicochemical and biological characteristics of the receiving soil [18,19]. Soil texture, pH, organic matter content and nutrient status, among other qualities, influence nutrient retention and transport processes, controlling nutrient mobility and potential losses [15–18]. Thus, identical fertilisers may display markedly different nutrient loss patterns when applied to contrasting soils [15,18,20,21].

Understanding these interactions has become increasingly important in recent years due to the rising costs and supply uncertainties associated with mineral fertilisers, driven by energy market volatility and geopolitical factors affecting global nutrient supply chains [20,22]. Although numerous studies have investigated nutrient losses following the application of organic amendments and recycled fertilisers, comparative assessments across contrasting soil types remain limited. In particular, direct comparisons among recovered fertilisers differing markedly in nutrient form and concentration under comparable experimental conditions are still scarce.

Therefore, the aim of this study was to evaluate the leaching dynamics of NH_4^+-N, NO_3^--N and PO_4^{3-}-P following the application of pig slurry, digestate and digestate-derived ammonium sulphate in agricultural soils with contrasting physicochemical characteristics. We hypothesized that nutrient leaching patterns would differ among fertiliser products due to their contrasting nutrient forms and release mechanisms, and that soil properties would play a key role in determining nutrient mobility and loss pathways.

2. Materials and Methods

A schematic representation of the experimental procedure is presented in Figure 2, illustrating the main stages of the study.

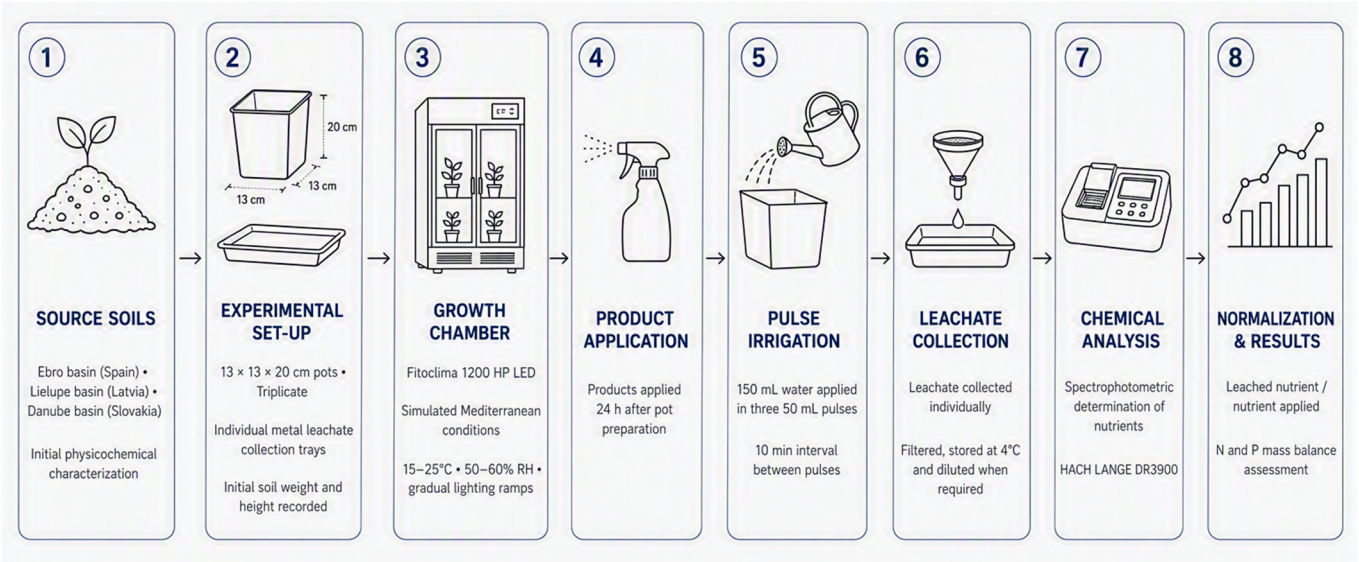


Figure 2. Schematic representation of the experimental workflow followed in the study.

2.1. Experimental Setup

The trials were carried out in commercial plastic pots (13 × 13 × 20 cm) filled to a soil depth of between 16 and 17 cm, corresponding to approximately 1.5–2.5 kg of soil per pot. This soil depth was selected to represent the upper soil layer where fertilisers are commonly applied and where most nutrient transformation and leaching processes occur immediately after fertilisation. The experimental design consisted of four soils, three fertiliser treatments and three replicates per treatment, resulting in a total of 36 experimental units.

Each pot was placed over an individual tray to allow independent leachate collection. The trials were carried out in a Fitoclima 1200 HP LED growth chamber (ARALAB, Río de Mouro, Portugal) which allowed precise regulation of radiation, air temperature and relative humidity, with gradual environmental ramps to avoid abrupt changes in climatic conditions [23]. Environmental conditions were selected to reproduce stable conditions representative of Southern European agricultural systems, given the strong influence of temperature and humidity on N transformations (nitrification, ammonification) and P mobility. The growth chamber followed a programmed 24 h cycle including a 16 h photoperiod and an 8 h dark period. Temperature ranged from 15 to 25 °C, relative humidity from 50 to 60%, and illumination from 0 to 60% of the maximum LED intensity.

The distance between the LED source and the support grid for the pots was 53 cm, with a distance to the soil surface of 20–27.5 cm. To minimise positional effects within the chamber, one replicate of each treatment was placed on each bench. The experimental matrix combined the application of the same fertilisers across different soil types.

2.2. Soils

The experimental trials were carried out using representative agricultural soils from three river basins with markedly different soil and climatic characteristics: the Ebro basin (Spain), the Lielupe basin (Latvia) and the Danube basin (Slovakia). The selected soils were chosen to represent contrasting physicochemical conditions across different European agricultural regions participating in the NENUPHAR project (NENUPHAR (New governance models to enhance nutrient pollution handling and nutrients recycling), Horizon Europe Grant Agreement No. 101082169. Project website: https://project-nenuphar.eu/). These soils covered a wide range of pH, texture, organic matter content and nutrient availability, allowing the influence of contrasting soil properties on nutrient mobility to be evaluated.

For each experimental campaign, a single batch of homogenized soil was used to fill all the pots of the same soil type, minimizing variability within each trial and ensuring comparability among the fertilisation treatments. The physicochemical properties of the soils are summarized in Table 1. Since the availability of analytical data differed among locations, the reported parameters vary accordingly. The soil characteristics before and after trials are presented in Appendix A.

Table 1. Characterization of soils used in the experiments.

	Orchard (Ebro Basin Demo)	La Melusa (Ebro Basin Demo)	Latvia (Lielupe Basin Demo)	Slovakia (Danube Basin Demo)
pH	8	8.12	7.30	6.82
EC (dS/m)	1.47	2.91	n.a.	n.a.
N total (%)	0.26	0.13	0.18	0.16
Organic matter (%)	3.52	1.79	2.80	2.67
Organic carbon (%)	2.04	1.04	1.62	1.55
C/N	7.8	8	9	9.7
Available P (mg/Kg)	100.1	12.73	37.93	48.30
Soil texture	Sandy clay loam	Sandy loam	Sandy loam	Clay loam

n.a., not available.

2.3. Tested Products and Application Rates

Three fertilisers representative of relevant agricultural waste streams were assessed: pig slurry, digestate and ammonium sulphate derived from digestate stripping. These products were selected because they originate from some of the largest-volume organic waste streams generated in Europe and, more specifically, in the Ebro basin, one of the most intensive livestock production regions on the continent. In line with the objectives of the NENUPHAR project, these waste-derived products were evaluated as nutrient recycling alternatives to support circular nutrient management. The physicochemical characterization of the products tested is presented in Table 2. Application volumes were calculated based on the nutrient concentrations of each fertiliser and application rates implemented at the NENUPHAR demonstration sites. The experiment simulated a single fertiliser application event representative of practical nutrient-recycling scenarios. Consequently, application volumes differed among products according to their nutrient concentrations, resulting in different nutrient inputs that reflected realistic field applications of pig slurry, digestate and ammonium sulphate. Nutrient loss results were subsequently normalised to the amount of nutrient applied in each treatment (Section 2.6), allowing for direct comparison of nutrient mobility among products with contrasting composition and agronomic application rates.

Table 2. Main characteristics of the tested products and estimated nutrient inputs applied per pot in the laboratory assays.

Product	Quantity/Pot (mL)	Dry Matter (%)	Total N Value (% DM)	Total P Value (% DM)	N Applied (mg)	P Applied (mg)
Pig slurry	150	0.5	19.70	1.51	147.75	11.30
Digestate	200	3.51	9.5	4.63	666.9	325.03
Ammonium sulphate	0.5	n.a.	9	n.a.	45	n.a.

DM, dry matter; n.a., not available.

Prior to application, all liquid products were thoroughly homogenised to ensure representative nutrient distribution. Fertilisers were applied 24 h after pot placement in the growth chamber to allow soil stabilization under controlled environmental conditions. The leaching assays were conducted on bare soil to quantify NH_4^+-N, NO_3^--N and PO_4^{3-}-P losses associated with each fertiliser treatment.

2.4. Irrigation and Leachate Collection

The irrigation was applied using distilled water in short pulses of water, instead of a single continuous volume, in order to promote a gradual infiltration, limit the preferential flow and runoff, and better reproduce the natural rainfall pattern in controlled conditions. Each irrigation event consisted of 150 mL applied in three pulses of 50 mL at 10 min intervals.

Irrigation events were performed at 24, 30, 144, 216 and 312 h after fertiliser application. The irrigation schedule was designed to capture both the short-term leaching response immediately after fertiliser application and the long-term dynamics associated with nutrient transformation in soil. The first two irrigation events (24 and 30 h) were closely spaced to simulate an early rainfall shortly after fertilisation, when the risk of NH_4^+-N and PO_4^{3-}-P leaching is highest, whereas the subsequent events (144, 216 and 312 h) were spaced at longer, regular intervals to monitor the progressive leaching of NO_3^--N as nitrification proceeded and to reflect a realistic sequence of discrete rainfall events over the following weeks. Following each irrigation event, pots were allowed to drain for 1 h to ensure complete leachate collection. Subsequently, the leachates were collected whenever present, and the total volume was measured and recorded.

Leachate samples were filtered, transferred into labelled containers and stored at 4 °C until chemical analysis. When the collected volume was insufficient for analytical determination, samples were diluted to the minimum volume required for analysis. The corresponding dilution factor was recorded and subsequently considered during data processing.

2.5. Chemical Analysis

The concentrations of nitrate (NO_3^--N), ammonium (NH_4^+-N) and phosphate (PO_4^{3-}-P) in the collected leachates were determined using a spectrophotometer DR3900 (HACH LANGE GmbH, Düsseldorf, Germany), using commercial HACH test kits according to the manufacturer's instructions.

Different analytical ranges were selected depending on the expected nutrient concentrations. Ammonium was determined using LCK305 (1.0–12.0 mg NH4-N L−1), LCK303 (2.0–47.0 mg NH4-N L−1) and LCK503 (10–100 mg NH4-N L−1) tests DR3900 (HACH LANGE GmbH, Düsseldorf, Germany). Nitrate concentrations were determined using LCK339 (0.23–13.5 mg NO_3-N L^{-1}) and LCK340 (5–35 mg NO_3-N L^{-1}) tests DR3900 (HACH LANGE GmbH, Düsseldorf, Germany). Orthophosphate concentrations were determined using LCK549 (0.01–0.5 mg PO_4-P L^{-1}) and LCK049 (0.05–1.5 mg PO_4-P L^{-1}) tests DR3900 (HACH LANGE GmbH, Düsseldorf, Germany).

Whenever nutrient concentrations exceeded the analytical range of the selected cuvette test, samples were diluted with distilled water and re-analysed using the appropriate analytical range. The corresponding dilution factor was subsequently applied to calculate the final nutrient concentration. Similarly, when the volume of leachate collected was insufficient for analytical determination, samples were diluted to the minimum volume required for analysis and the corresponding dilution factor was accounted for during data processing.

2.6. Normalization

Since application rates were not equalized between treatments, absolute nutrient concentrations in the leachates could not be directly compared across products. Nutrient losses were normalized to the amount of nutrient applied in each treatment according to the following Equation (1):

$$NL_i = \frac{M_{i,leached}}{M_{i,applied}} \quad (1)$$

where NL_i is the normalized loss of nutrient i (NH_4^+-N, NO_3^--N or PO_4-P), $M_{i,leached}$ is the cumulative mass of nutrient i recovered in the leachates (mg), and $M_{i,applied}$ is the mass of nutrient i applied per pot (mg).

This normalization facilitated comparisons among products despite differences in nutrient content and application rates. All results are presented as the mean of three independent replicates.

2.7. Statistical Analysis

Each soil–fertiliser combination comprised three independent experimental replicates. Nutrient losses measured at each sampling event were normalized to the amount of N or P applied, allowing for direct comparisons among fertiliser treatments despite differences in nutrient concentration and application rates.

Temporal leaching data were used to describe nutrient-loss dynamics throughout the experimental period and are presented in Figure 3. Variability among replicates is reported in Table 3 as mean ± standard error (SE) of the three independent replicates.

Table 3. Cumulative normalized nutrient losses (mean ± SE, n = 3). Mineral N represents NH_4^+-N + NO_3^--N. Normalized losses were calculated as cumulative nutrient mass in leachates divided by nutrient input per pot. Phosphate losses were not calculated for ammonium sulphate because no P was applied. n.a., not applicable. Statistical effects of soil, fertiliser and their interaction were evaluated by two-way ANOVA, followed by Tukey-adjusted comparisons among fertilisers within each soil ($p < 0.05$).

Soil	Fertiliser	NH_4-N/N Applied	NO_3-N/N Applied	Mineral N/N Applied	PO_4-P/P Applied
La Melusa	Pig slurry	0.00219 ± 0.00034	0.00442 ± 0.00132	0.00661 ± 0.00109	0.000450 ± 0.000120
	Digestate	0.00104 ± 0.00038	0.00189 ± 0.00146	0.00293 ± 0.00181	0.000007 ± 0.000002
	Ammonium sulphate	0.11249 ± 0.04107	0.01210 ± 0.00584	0.12459 ± 0.04628	n.a.
Orchard	Pig slurry	0.04898 ± 0.00171	0.12708 ± 0.01837	0.17606 ± 0.01903	0.037312 ± 0.009816
	Digestate	0.03686 ± 0.00855	0.04263 ± 0.00971	0.07948 ± 0.01815	0.002693 ± 0.001007
	Ammonium sulphate	0.05934 ± 0.01661	0.63171 ± 0.09443	0.69105 ± 0.10576	n.a.
Latvia	Pig slurry	0.03491 ± 0.00322	0.01222 ± 0.00297	0.04713 ± 0.00500	0.123141 ± 0.015512
	Digestate	0.01281 ± 0.00289	0.00314 ± 0.00020	0.01595 ± 0.00306	0.003765 ± 0.001039
	Ammonium sulphate	0.01156 ± 0.00466	0.04810 ± 0.01034	0.05966 ± 0.01041	n.a.
Slovakia	Pig slurry	0.00108 ± 0.00037	0.01340 ± 0.00204	0.01448 ± 0.00241	0.001455 ± 0.000647
	Digestate	0.00576 ± 0.00133	0.00764 ± 0.00047	0.01340 ± 0.00181	0.004957 ± 0.000201
	Ammonium sulphate	0.00042 ± 0.00006	0.02773 ± 0.00386	0.02816 ± 0.00392	n.a.

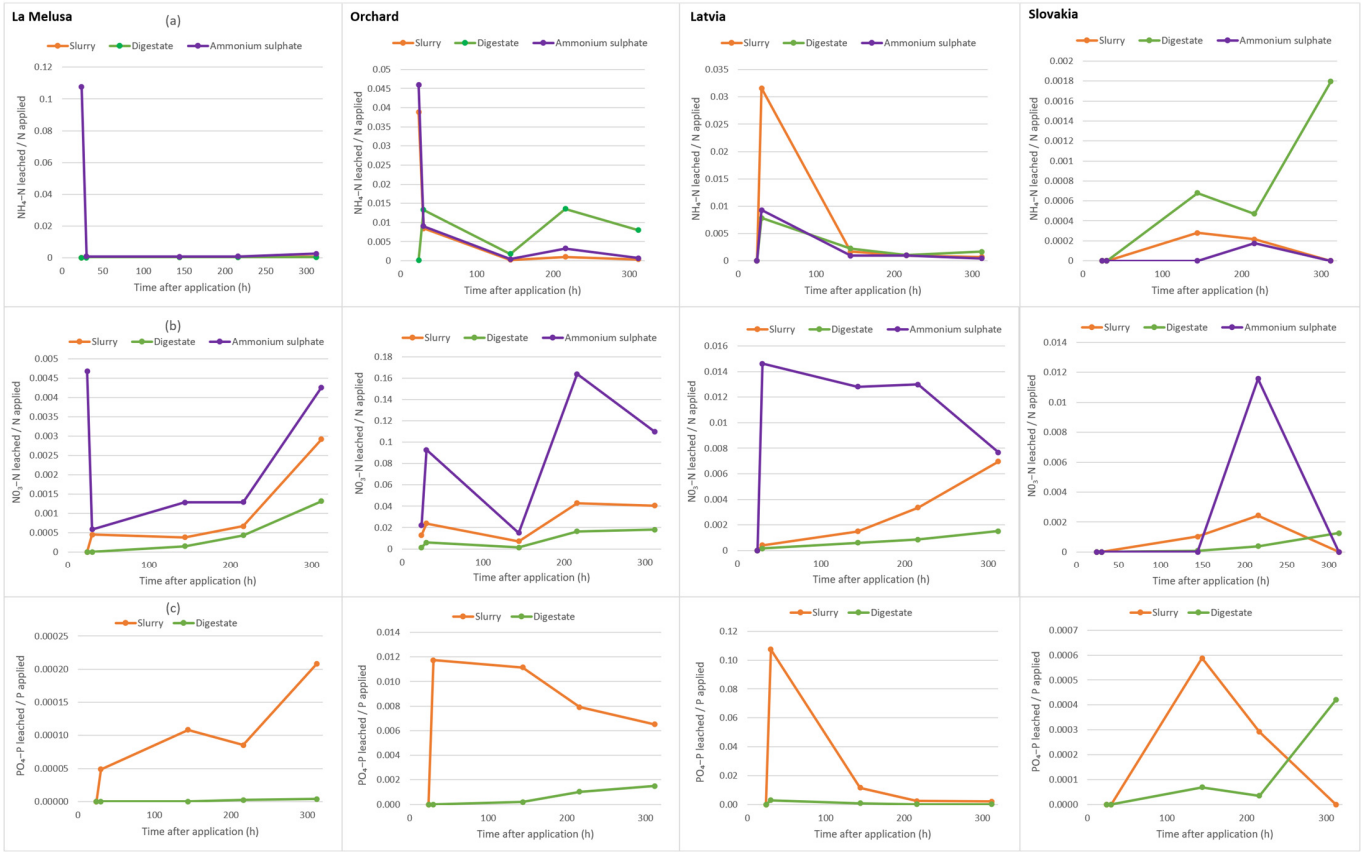


Figure 3. Temporal evolution of normalized nutrient leaching following the application of pig slurry, digestate and ammonium sulphate in four contrasting agricultural soils. Panels show (**a**) NH_4^+-N, (**b**) NO_3^--N and (**c**) PO_4^{3-}-P losses normalized to the amount of nutrient applied. For clarity, error bars are not displayed because the inclusion of replicate variability substantially reduced the readability of the multi-panel time-series figure. Variability among replicates is reported in Table 3 as mean ± standard error (SE; $n = 3$), together with the corresponding statistical analyses.

For each replicate, cumulative normalized nutrient losses were calculated by summing nutrient losses measured at all sampling times (24, 30, 144, 216 and 312 h after fertiliser application). The effects of soil, fertiliser and their interaction on cumulative normalized NH_4^+-N, NO_3^--N, mineral N and PO_4^{3-}-P losses were evaluated using two-way analysis of variance (ANOVA). Phosphorus analyses excluded ammonium sulphate because this fertiliser did not contain phosphorus.

When significant effects were detected, Tukey's honestly significant difference (HSD) test was used for pairwise comparisons among treatments. Statistical significance was established at $p < 0.05$. Results are reported as mean ± SE unless otherwise indicated.

3. Results

Differences in leachate generation were observed among soils during the experimental period, resulting in distinct nutrient loss patterns. La Melusa showed a limited and discontinuous leaching response, whereas Orchard and Latvia generated more consistent leachate volumes throughout the experiment. Detailed nitrogen and phosphorus balances are provided in Appendix A (Tables A1–A8).

Two-way ANOVA revealed significant effects of soil, fertiliser and their interaction on normalized NH_4^+-N, NO_3^--N, mineral N and PO_4^{3-}-P losses ($p < 0.05$), indicating that nutrient-loss responses depended on both fertiliser type and the receiving soil. Tukey-adjusted pairwise comparisons further confirmed significant differences among fertilisers

within several soil–nutrient combinations ($p < 0.05$). Mainly, treatments associated with greater nitrate losses tended to involve ammonium sulphate, whereas pig slurry frequently generated higher phosphate losses. The magnitude and significance of these differences varied among soils, reflecting the strong influence of soil properties on nutrient mobility and retention.

The temporal evolution of ammonium (NH_4^+-N), nitrate (NO_3^--N) and phosphate (PO_4^{3-}-P) leaching, normalized to the amount of nutrient applied, is presented in Figure 3.

3.1. Temporal Evolution of Nutrient Leaching

The temporal evolution of nutrient leaching differed among fertilisers and soils (Figure 3). Ammonium sulphate produced the highest NH_4^+-N losses during the initial sampling period. This effect was particularly evident in La Melusa and Orchard soils, where the largest NH_4^+-N values were recorded during the first sampling event. In contrast, pig slurry and digestate showed lower NH_4^+-N losses distributed over a longer period.

The evolution of NO_3^--N losses varied more markedly among soils. Orchard presented the highest nitrate losses, particularly following ammonium sulphate application. Latvia also showed measurable NO_3^--N losses, although differences among fertilisers were less pronounced. In La Melusa and Slovakia, nitrate losses remained comparatively lower throughout the experimental period. The relative contribution of NH_4^+-N and NO_3^--N differed depending on both soil and fertiliser type.

Phosphate losses remained comparatively low across most soil–fertiliser combinations. Pig slurry produced the highest normalized phosphate losses in La Melusa, Orchard and Latvia soils. Digestate and ammonium sulphate exhibited limited phosphate leaching during most of the experimental period. In Slovakia, an increase in PO_4^{3-}-P losses was observed for digestate during the final sampling event.

Differences were also observed in the timing of nutrient release. While NH_4^+-N losses were concentrated during the first stages of the experiment, NO_3^--N and PO_4^{3-}-P losses often followed a more gradual pattern, with maximum values occurring at later sampling times depending on the soil-fertiliser combination.

3.2. Cumulative Nutrient Losses

Cumulative nutrient losses measured over the complete experimental period and normalized to the corresponding amount of N or P applied are presented in Table 3. Considerable variability was observed among soil–fertiliser combinations for all nutrient forms evaluated.

The highest normalized N loss was recorded for ammonium sulphate in Orchard, followed by pig slurry in Orchard and ammonium sulphate in La Melusa. In contrast, the lowest mineral N losses were observed in La Melusa for digestate and in Slovakia for digestate and pig slurry.

Differences were also identified in the relative contribution of NH_4^+-N and NO_3^--N to cumulative mineral N losses. The highest normalized NH_4^+-N loss was observed for ammonium sulphate in La Melusa, whereas the highest normalized NO_3^--N loss was recorded for ammonium sulphate in Orchard. In Orchard, NO_3^--N accounted for most of the mineral recovered in the leachates for all fertilisers. In Slovakia, NO_3^--N also represented the dominant N form in the leachates, despite the lower magnitude of cumulative N losses compared with Orchard.

Normalized phosphate losses showed a different pattern. Pig slurry produced the highest PO_4-P losses in Latvia and Orchard, whereas digestate exhibited its maximum normalized PO_4-P loss in Slovakia. Digestate produced lower normalized phosphate losses

than pig slurry in La Melusa, Orchard and Latvia. No consistent ranking between both products was observed across the four soils.

The cumulative loss data indicate that nutrient mobility differed among receiving soils and fertiliser types. The ranking of nutrient losses changed according to both the nutrient form and the fertiliser considered, revealing distinct responses among soil–fertiliser combinations.

3.3. General Trends in Nutrient Losses

The leaching patterns observed in Figure 3 did not show a uniform response for any of the fertilisers across all soils. The magnitude of nutrient losses, the dominant N form detected in the leachates and the temporal evolution of nutrient release varied among locations for the same fertiliser product.

Differences among soils were particularly evident for NO_3^--N losses. For example, Orchard and Latvia showed higher relative nitrate losses than La Melusa and Slovakia, whereas the contribution of NH_4^+-N to total N losses differed depending on the soil-fertiliser combination. Similar differences were observed for phosphate, with contrasting leaching responses among soils despite the application of the same products.

The behaviour of pig slurry and digestate was not consistent across all locations. In some soils, these products generated comparatively low nutrient losses, whereas in others they were associated with higher NH_4^+-N, NO_3^--N or PO_4^{3-}-P leaching. Likewise, although ammonium sulphate frequently showed higher NH_4^+-N losses immediately after application, this trend was not systematically observed for the other nutrient forms.

The data show that nutrient losses depended on both the fertiliser applied and the soil under study. No fertiliser showed consistently lower or higher leaching losses across all soils, and the response to fertiliser application varied according to the soil-fertiliser combination evaluated.

3.4. Relative Comparison of Soil Properties and Nutrient Losses

In order to facilitate the comparison of soil characteristics and nutrient leaching responses among treatments, radar plots were constructed for each fertiliser type (Figure 4). The plots combine selected soil properties (sand content, clay content, organic matter, pH, total N and available P) with cumulative normalized nutrient losses. Since the variables were expressed in different units and ranges, each variable was independently rescaled between 0 and 1 using min–max normalization within each fertiliser treatment. The radar plots only represent relative differences among soils within a given fertiliser treatment and do not allow direct quantitative comparisons between fertilisers.

The radar plots show that nutrient-loss patterns varied among soils within each fertiliser treatment. The relative position of each soil changed depending on the fertiliser and nutrient considered, indicating that the response to fertilisation was not uniform across the evaluated soil–fertiliser combinations. The relative ranking of soils varied according to both nutrient form and fertiliser type, indicating that no individual soil property showed a consistent relationship with nutrient losses across all treatments.

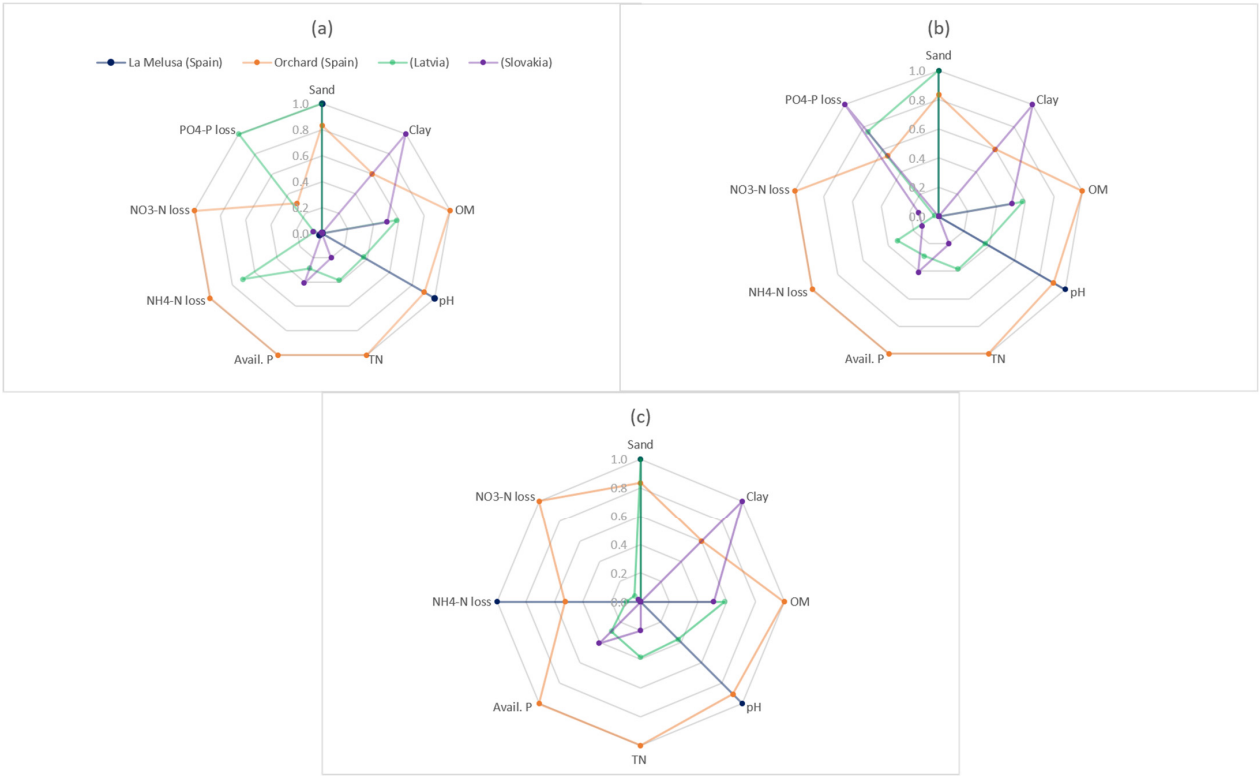


Figure 4. Radar plots showing the relative profiles of selected soil properties and normalized nutrient losses for (**a**) pig slurry, (**b**) digestate and (**c**) ammonium sulphate. Variables were independently rescaled between 0 and 1 using min–max normalization within each fertiliser treatment. Values of 1 and 0 correspond to the highest and lowest observed values, respectively, for each variable within the corresponding fertiliser treatment. The variable PO_4-P loss was only included for pig slurry and digestate because no phosphorus was applied with ammonium sulphate. Abbreviations: NH_4-N, ammonium nitrogen; NO_3-N, nitrate nitrogen; PO_4-P, phosphate phosphorus; OM, organic matter; TN, total nitrogen; Avail. P, available phosphorus.

4. Discussion

4.1. Influence of Fertiliser Type on Nitrogen Leaching Dynamics

The three fertilisers evaluated represent contrasting nitrogen sources, differing in nutrient concentration and nitrogen speciation. These differences were also evident in the leaching patterns observed during the experiment.

Ammonium sulphate produced the highest NH_4^+-N losses during the first sampling events. This behaviour is consistent with the chemical nature of the product, which contains nitrogen predominantly in mineral ammonium form and becomes readily available following dissolution in soil water. Recent evidence on recovered fertilisers similarly indicates that ammonium sulphate obtained through stripping-scrubbing provides a highly soluble and readily available source of ammoniacal N, although its subsequent fate remains dependent on soil conditions and application management [14].

In contrast, pig slurry and digestate showed a more gradual NH_4^+-N release pattern. These products contain both mineral and organic nitrogen fractions, and part of the applied N must undergo mineralization before becoming available in inorganic forms, delaying its mobilization within the soil profile [21,24].

The temporal evolution of NH_4^+-N and NO_3^--N losses also revealed differences among nitrogen sources. In several treatments, nitrate losses became more relevant after the initial ammonium peak. The balance between NH_4^+-N and NO_3^--N in soil is highly variable and is influenced by soil texture, pH, organic matter and management conditions,

which regulate nitrification and the retention or transport of mineral N forms [25]. This pattern is consistent with the transformation of ammonium into nitrate following fertiliser application. Since nitrate remains largely in the soil solution and is not retained by negatively charged soil particles, it is more susceptible to transport with percolating water than ammonium [26,27]. This behaviour is consistent with the higher contribution of nitrate to total N losses observed in several soil-fertiliser combinations.

However, fertiliser behaviour was not consistent across all soils. Ammonium sulphate generated the highest relative NH_4^+-N losses in most soils, but this pattern was not observed in Slovakia, where lower ammonium losses were recorded. Likewise, differences among pig slurry, digestate and ammonium sulphate varied depending on the receiving soil, particularly for nitrate leaching. The results show that fertiliser composition alone was insufficient to explain nitrogen losses and suggest that soil conditions strongly influenced the fate of applied nitrogen. A plausible explanation for the anomalous pattern observed in Slovakia relates to its soil properties: this soil combined the lowest pH (6.82) and the finest texture (clay loam) of the four soils (Table 1). The higher clay content likely provided greater cation exchange capacity, favouring electrostatic retention of NH_4^+-N on negatively charged clay surfaces, and potentially additional non-exchangeable fixation of NH_4^+-N within clay interlayers, a mechanism well documented for illicit and vermiculitic clay minerals [28,29]. Once fixed, NH_4^+-N is no longer available for nitrification, which directly limits its subsequent conversion to the more mobile NO_3^--N form [28]. The comparatively lower soil pH may also have slowed nitrification rates directly, as soil pH is recognised as one of the main factors controlling nitrification [30], further prolonging the residence time of NH_4^+-N in the soil matrix and increasing its exposure to these retention mechanisms rather than being lost with percolating water. Together, these soil-specific properties may have limited NH_4^+-N leaching from ammonium sulphate in Slovakia, despite this fertilizer supplying readily available inorganic NH_4^+ from the outset. The soil-dependent differences among fertilisers in nitrate leaching likely reflect the interaction between each product's N release pattern and the receiving soil's nitrification capacity and hydraulic behaviour. Ammonium sulphate provides an immediate pulse of inorganic NH_4^+-N, so its conversion to NO_3^--N, and subsequent leaching, is governor primarily by the rate of nitrification in each soil. By contrast, pig slurry and digestate contain a substantial organic N fraction that must first undergo mineralisation, staggering N release and potentially desynchronising nitrate production from leaching events in soils with slower N transformation or higher retention capacity. As nitrification rates and cation exchange capacity were not directly measured in this study, these interpretations remain inferential and would benefit from targeted measurements in future work.

The differences observed between pig slurry and digestate were smaller than those observed between these products and ammonium sulphate. Although anaerobic [24,31] digestion increases the proportion of ammoniacal nitrogen compared with untreated livestock residues, digestate still contains a substantial organic N fraction and retains some of the release characteristics associated with organic fertilisers [24,31]. This intermediate composition may explain why digestate frequently showed leaching responses between those observed for ammonium sulphate and pig slurry.

4.2. Phosphorus Behaviour

Phosphate losses remained low across the evaluated soil–fertiliser combinations, reflecting the lower mobility of phosphorus compared with mineral N forms.

Unlike nitrogen, no consistent ranking was observed between pig slurry and digestate across the evaluated soils. Pig slurry generated the highest relative phosphorus losses in Latvia and Orchard, whereas digestate showed its highest losses in Slovakia. These

contrasting responses indicate that phosphorus mobility was influenced by both fertiliser characteristics and soil properties, with the relative importance of each factor varying among locations [32,33].

The different behaviour observed for pig slurry and digestate may be related to differences in the chemical forms in which phosphorus is present in each product, as reported in previous studies. Anaerobic digestion can modify phosphorus distribution through changes in pH and the formation of mineral precipitates containing calcium, magnesium or ammonium, potentially modifying phosphorus availability and mobility compared with untreated livestock residues [24,34,35]. Nevertheless, phosphorus speciation was not determined in the present study and the mechanisms responsible for the observed differences cannot be directly confirmed.

The absence of a common phosphorus loss pattern across soils further indicates that phosphorus mobility cannot be attributed solely to fertiliser composition. The response depended on the combination of fertiliser type and receiving soil, resulting in contrasting phosphorus losses among experimental sites.

4.3. Influence of Soil Properties on Nutrient Losses

The radar plots showed that none of the fertilisers followed a uniform leaching pattern across the four soils (Figure 4). Differences among soils were more evident for NO_3^--N and PO_4^{3-}-P, whereas the early NH_4^+-N response was more closely related to fertiliser type. The soil associated with the highest nutrient loss also varied according to the fertiliser and nutrient form considered. The observed leaching patterns could not be attributed to a single soil property.

Soil texture influences water movement through the soil profile, while clay and organic matter affect water retention and the retention of positively charged nutrients. Recent evidence indicates that N-leaching vulnerability is governed by the interaction between soil hydraulic properties, water fluxes and the characteristics of the N source, rather than by texture alone [36]. These properties are particularly relevant to the different behaviour of ammonium and nitrate. Ammonium can be retained on negatively charged soil surfaces, whereas nitrate remains mainly in the soil solution and is more readily transported with percolating water [17,26,35]. Nevertheless, the radar plots did not show a consistent correspondence between texture or organic matter content and N losses across all fertiliser treatments.

Orchard, which had the highest organic matter and total N contents, showed the highest relative NO_3^--N losses for the three fertilisers. In contrast, La Melusa presented limited leaching and low nutrient losses, despite having a pH similar to that of Orchard. The different responses of these two Ebro soils indicate that pH or organic matter content alone did not account for the observed N leaching patterns. A possible explanation for this contrasting behaviour lies in the different roles that organic matter and total N content play in nitrate dynamics compared with ammonium retention. Because NO_3^--N is a mobile anion that is not appreciably retained by the predominantly negatively charged surfaces of clay minerals and organic matter [18], its leaching is governed primarily by the rate at which NH_4^+-N is nitrified, a process strongly controlled by substrate availability and soil pH [30], and by the soil's hydraulic behaviour, rather than by sorption capacity. Orchard's higher organic matter and total N content likely sustained a larger, more active pool of mineralisable N, fuelling faster nitrification and generating a greater supply of NO_3^--N available for transport with percolating water. In contrast, La Melusa's lower organic matter and total N content would be expected to support comparatively lower nitrification rates, limiting NO_3^--N generation despite its similar pH. Differences in soil structure between the two Ebro soils (Table 1) may also have contributed, with Orchard's finer

texture (sandy clay loam vs. sandy loam) potentially favouring structural macropores that allow preferential, rapid NO_3^--N transport bypassing matrix retention [37]. As nitrification rates and hydraulic conductivity were not directly measured in this study, further research combining these measurements would be needed to confirm the relative contribution of each mechanism. Comparable assessments of soil mineral N dynamics have also shown that the balance between nitrate and ammonium reflects interacting soil factors rather than a single physicochemical variable [25]. The differences may be related to the combined effects of soil properties on water movement, nutrient retention and N transformation [38]. Studies conducted in temperate European soils have similarly shown that N mineralisation and availability cannot be reliably predicted from total organic carbon or total N alone, and that combining mineralisation indicators with pH, carbon-to-nitrogen relationships and clay-related properties improves the interpretation of soil N dynamics [39].

Phosphorus losses also varied among soils without following a consistent trend based on pH, texture or initial available P. Orchard had the highest available P content, but the highest relative PO_4^{3-}-P loss following pig slurry application was recorded in Latvia, while digestate produced its highest relative PO_4^{3-}-P loss in Slovakia. Phosphorus mobility is controlled by concurrent adsorption, desorption and precipitation processes, which are affected by soil pH, mineral composition, organic matter and initial P status [19,35,40]. The contrasting rankings obtained for pig slurry and digestate indicate that soil retention processes acted in combination with the characteristics of the applied product.

The results support the initial hypothesis that nutrient leaching depends on both fertiliser composition and the characteristics of the receiving soil. Fertiliser composition influenced the form and initial availability of the applied nutrients, while soil conditions affected their retention, transformation and transport through the soil profile. As a result, no fertiliser could be assigned a uniform leaching risk across the soils evaluated. The environmental performance of recovered fertilisers should be assessed in relation to the receiving soil rather than considered an intrinsic and constant property of the product.

5. Conclusions

This study evaluated the leaching behaviour of nitrogen and phosphorus following the application of pig slurry, digestate and digestate-derived ammonium sulphate in four agricultural soils with contrasting properties. The results showed differences among soil–product combinations, indicating that nutrient losses are strongly influenced by both fertiliser type and soil characteristics.

Nitrogen losses showed greater variability than phosphorus losses, with contrasting contributions of NH_4^+-N and NO_3^--N depending on the receiving soil. In contrast, phosphorus leaching remained low under the tested conditions. These findings support the consideration of soil-specific conditions when assessing the environmental performance of recovered fertilisers.

Although the experiments were conducted under controlled conditions, they provide valuable insights into the behaviour of recovered nutrient products in different soil systems. However, several limitations of this study should be acknowledged. First, the leaching assays were conducted on bare soil, without actively growing plants; consequently, the nutrient losses reported here likely represent a worst-case scenario relative to cropped systems, where plant uptake would compete with leaching and could substantially reduce N and P losses. Second, the experimental period (312 h) does not reflect the seasonal climatic variability, multiple applications, or extended crop cycles typical of field conditions. Finally, although four soils from three contrasting European regions were included, this represents a limited subset of European soil and climatic diversity, and the controlled climatic regime does not reproduce the intensity or variability of natural rainfall events, including extreme

precipitation associated with climate change. Further research under field conditions is needed to confirm these trends and to improve our understanding of the interactions between fertiliser properties, soil characteristics and nutrient losses.

Soil properties exerted strong control on nutrient mobility and frequently modified the response expected from fertiliser composition alone. Consequently, the environmental performance of recovered fertilisers should be assessed in relation to the receiving soil rather than considered an intrinsic property of the fertiliser itself.

Author Contributions: Conceptualization, B.P. and P.d.l.S.; methodology, B.P., P.d.l.S. and T.D.M.; validation, B.P., P.d.l.S. and T.D.M.; formal analysis, B.P.; investigation, B.P., P.d.l.S. and T.D.M.; data curation, B.P.; writing—original draft preparation, B.P., P.d.l.S. and T.D.M.; writing—review and editing, B.P. and P.d.l.S.; visualization, B.P. All authors have read and agreed to the published version of the manuscript.

Funding: This work was supported by the European Commission under the NENUPHAR Project with Grant Agreement No. 101082169. The views and opinions expressed are those of the authors only and do not necessarily reflect those of the European Union or the European Commission. Neither the European Union nor the granting authority can be held responsible for them.

Data Availability Statement: The main data supporting the findings of this study are included within the article and its Appendix A. Additional data is available from the corresponding author upon reasonable request.

Acknowledgments: The authors would like to thank the NENUPHAR project partners for providing the soil and nutrient recovery products used in this study. During the preparation of this manuscript, the authors used Microsoft 365 Copilot (Microsoft Corporation, Redmond, WA, USA; cloud-based generative AI assistant integrated within Microsoft 365, accessed in August 2026) to support language editing, text structuring, and the presentation of calculations and tables. The authors reviewed and edited all generated outputs and take full responsibility for the scientific content of this publication.

Conflicts of Interest: The authors declare no conflicts of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript; or in the decision to publish the results.

Abbreviations

The following abbreviations are used in this manuscript:

C/N	Carbon-to-nitrogen ratio
DM	Dry matter
EC	Electrical conductivity
N	Nitrogen
NH_4^+-N	Ammonium nitrogen
NO_3^--N	Nitrate nitrogen
OM	Organic matter
P	Phosphorus
PO_4^--P	Phosphate phosphorus
TN	Total Nitrogen
n.a.	Not available/not applicable

Appendix A

This appendix provides the complete nitrogen and phosphorus balances obtained for each soil–fertiliser combination evaluated in the experiment. The tables include nutrient inputs, initial and final soil nutrient contents, net nutrient accumulation within the soil system and cumulative nutrient losses measured in the leachates. These supplementary

data support the normalized nutrient loss indicators presented in the main text and ensure full transparency and reproducibility of the calculations performed.

Table A1. Nitrogen balance in La Melusa soil.

	Inputs N (mg)	Soil System			Outputs N (Leachates)		
		Initial N (mg)	Final N (mg)	N Increased (mg)	NH_4-N (mg)	NO_3-N (mg)	N Total (mg)
Pig slurry	147.75	2823.38	3097.34	273.96	3.24×10^{-1}	6.53×10^{-1}	9.77×10^{-1}
Digestate	666.9	2573.25	3892.09	1318.85	6.90×10^{-1}	1.26	1.95
Ammonium sulphate	45	2823.38	2983.29	159.91	5.06	5.54×10^{-1}	5.61

Table A2. Phosphorus balance in La Melusa soil.

	Inputs P (mg)	Soil System			Outputs P (Leachates)
		Initial P (mg)	Final P (mg)	P Increased (mg)	PO_4-P (mg)
Pig slurry	11.3	27,114.90	42,955.83	15,840.93	5.09×10^{-3}
Digestate	325.03	24,445.84	115,921.22	91,475.37	2.12×10^{-3}
Ammonium sulphate	-	-	-	-	-

Table A3. Nitrogen balance in Orchard soil.

	Inputs N (mg)	Soil System			Outputs N (Leachates)		
		Initial N (mg)	Final N (mg)	N Increased (mg)	NH_4-N (mg)	NO_3-N (mg)	N Total (mg)
Pig slurry	147.75	4379.27	4634.14	254.87	7.24	$1.88 \times 10^{+1}$	$2.60 \times 10^{+1}$
Digestate	666.9	4368.87	5045.45	676.58	$2.46 \times 10^{+1}$	$2.84 \times 10^{+1}$	$5.30 \times 10^{+1}$
Ammonium sulphate	45	4236.27	4221.45	−14.81	2.67	$1.82 \times 10^{+1}$	$2.08 \times 10^{+1}$

Table A4. Phosphorus balance in Orchard soil.

	Inputs P (mg)	Soil System			Outputs P (Leachates)
		Initial P (mg)	Final P (mg)	P Increased (mg)	PO_4-P (mg)
Pig slurry	11.3	168.60	180.38	11.78	4.21×10^{-1}
Digestate	325.03	168.20	188.57	20.37	8.75×10^{-1}
Ammonium sulphate	-	-	-	-	-

Table A5. Nitrogen balance in Latvia soil.

	Inputs N (mg)	Soil System			Outputs N (Leachates)		
		Initial N (mg)	Final N (mg)	N Increased (mg)	NH_4-N (mg)	NO_3-N (mg)	N Total (mg)
Pig slurry	147.75	2915.94	3423.00	507.06	5.16	1.81	6.96
Digestate	666.9	2927.18	3863.10	935.92	8.54	2.10	$1.06 \times 10^{+1}$
Ammonium sulphate	45	2787.26	2636.07	−151.18	5.20×10^{-1}	2.16	2.68

Table A6. Phosphorus balance in Latvia soil.

	Inputs P (mg)	Soil System			Outputs P (Leachates)
		Initial P (mg)	Final P (mg)	P Increased (mg)	PO_4-P (mg)
Pig slurry	11.295	59.02	147.97	88.95	1.39
Digestate	325.026	59.25	137.57	78.32	1.22
Ammonium sulphate	-	-	-	-	-

Table A7. Nitrogen balance in Slovakia soil.

	Inputs N (mg)	Soil System			Outputs N (Leachates)		
		Initial N (mg)	Final N (mg)	N Increased (mg)	NH_4-N (mg)	NO_3-N (mg)	N Total (mg)
Pig slurry	147.75	-	2574.82	2574.82	1.59×10^{-1}	1.98	2.14
Digestate	666.9	-	3408.28	3408.28	3.84	5.10	8.93
Ammonium sulphate	45	-	2210.56	2210.56	1.90×10^{-2}	1.25	1.27

Table A8. Phosphorus balance in Slovakia soil.

Inputs P (mg)		Soil System			Outputs P (Leachates)
Initial P (mg)	Final P (mg)	P Increased (mg)			PO_4-P (mg)
Pig slurry	11.295	81.35	102.64	21.29	1.64×10^{-2}
Digestate	325.026	81.16	132.61	51.45	1.61
Ammonium sulphate	-	-	-	-	-

References

1. European Commission. Soil Health. Available online: https://environment.ec.europa.eu/topics/soil-health_en (accessed on 9 August 2026).
2. Akca, E.; Aldrien, U.; Alewell, C. *The State of Soils in Europe*; Publications Office of the European Union: Luxembourg, 2024. [CrossRef]
3. European Environment Agency. *Soil Monitoring in Europe: Indicators and Thresholds for Soil Health Assessments*; European Environment Agency: Copenhagen, Denmark, 2024. [CrossRef]
4. FAO. *Status of the World's Soil Resources 2026*; Food and Agriculture Organization of the United Nations (FAO): Rome, Italy, 2026. [CrossRef]
5. Soares, M.; Genisheva, Z.; Miranda, T.; Pereira, E.; Vilarinho, C.; Carvalho, J. Agricultural Waste: The Picture of European Union Countries. *J. Sustain. Res.* **2025**, *7*, 3047. [CrossRef]
6. Grizzetti, B.; Vigiak, O.; Aguilera, E.; Aloe, A.; Biganzoli, F.; Billen, G.; Caldeira, C.; De, M.; Egle, L.; Einarsson, R.; et al. *Knowledge for Integrated Nutrient Management Action Plan (INMAP)*; Publications Office of the European Union: Luxembourg, 2023.
7. Moreno-Lora, A.; Sayadi-Gmada, S.; Fernández-Fernández, M.M.; Suárez-Rey, E.M. Nitrate Pollution of Water Bodies from Agricultural Sources: The Role of Training in Enhancing Awareness and Knowledge in Andalusia. *Nitrogen* **2025**, *6*, 103. [CrossRef]
8. European Parliament and Council of the European Union. Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 Laying Down Rules on the Making Available on the Market of EU Fertilising Products and Amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and Repealing Regulation (EC) No 2003/2003 (Text with EEA Relevance). *Off. J. Eur. Union* **2019**, *L170*, 1–114. Available online: http://data.europa.eu/eli/reg/2019/1009/oj (accessed on 9 August 2026).
9. Vaneeckhaute, C.; Lebuf, V.; Michels, E.; Belia, E.; Vanrolleghem, P.A.; Tack, F.M.; Meers, E. Nutrient Recovery from Digestate: Systematic Technology Review and Product Classification. *Waste Biomass Valor.* **2017**, *8*, 21–40. [CrossRef]
10. Huygens, D.; Delgado Sancho, L.; Saveyn, H.G.M.; Tonini, D.; Eder, P. *Technical Proposals for Selected New Fertilising Materials under the Fertilising Products Regulation (Regulation (EU) 2019/1009): Process and Quality Criteria, and Assessment of Environmental and Market Impacts for Precipitated Phosphate Salts & Derivates, Thermal Oxidation Materials & Derivates and Pyrolysis & Gasification Materials*; Publications Office of the European Union: Luxembourg, 2019.
11. Köninger, J.; Lugato, E.; Panagos, P.; Kochupillai, M.; Orgiazzi, A.; Briones, M.J.I. Manure management and soil biodiversity: Towards more sustainable food systems in the EU. *Agric. Syst.* **2021**, *194*, 103251. [CrossRef]
12. European Biogas Association (EBA). *EBA Statistical Report 2025*; European Biogas Association: Brussels, Belgium, 2025.
13. De Marco, T.; Dorado-Sánchez, C.; Carmona-Martínez, A.; Palacino-Blazquez, B.; Aragón-Briceño, C. Assessing the Economic Feasibility of Nitrogen and Phosphorus Recovery Systems in European Waste Valorization Case Studies. *Sustainability* **2026**, *18*, 7041. [CrossRef]
14. Shrivastava, V.; Laasri, I. Nutrient Recovery Strategies and Agronomic Performance in Circular Farming: A Comprehensive Review. *Nitrogen* **2025**, *6*, 80. [CrossRef]
15. Schimel, J.P.; Bennett, J. Nitrogen Mineralization: Challenges of a Changing Paradigm. *Ecology* **2004**, *85*, 591–602. [CrossRef]
16. Robertson, G.P.; Groffman, P.M. Nitrogen Transformations. In *Soil Microbiology, Ecology and Biochemistry*; Elsevier: Amsterdam, The Netherlands, 2024; pp. 407–438. [CrossRef]
17. Bruno, I.P.; Araújo, A.G.; Merten, G.H.; Ladeira, A.S.; Pinto, V.M. Crop Rotation and Nitrogen Fertilizer on Nitrate Leaching: Insights from a Low Rainfall Study. *Nitrogen* **2024**, *5*, 329–348. [CrossRef]

18. Cameron, K.C.; Di, H.J.; Moir, J.L. Nitrogen Losses from the Soil/Plant System: A Review. *Ann. Appl. Biol.* **2013**, *162*, 145–173. [CrossRef]
19. García-Albacete, M.; Tarquis, A.M.; Cartagena, M.C. Risk of Leaching in Soils Amended by Compost and Digestate from Municipal Solid Waste. *Sci. World J.* **2014**, *2014*, 565174. [CrossRef] [PubMed]
20. World Bank. *Commodity Markets Outlook, April 2022: The Impact of the War in Ukraine on Commodity Markets*; World Bank: Washington, DC, USA, 2022.
21. Niedziński, T.; Sierra, M.J.; Łabętowicz, J.; Noras, K.; Cabrales, C.; Millán, R. Release of Nitrogen from Granulate Mineral and Organic Fertilizers and Its Effect on Selected Chemical Parameters of Soil. *Agronomy* **2021**, *11*, 1981. [CrossRef]
22. European Commission. *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: Fertiliser Action Plan. Partnership for Ensuring the Availability, Affordability and Strategic Autonomy of Home-Grown EU Fertilisers*; European Commission: Brussels, Belgium, 2026.
23. Annunziata, M.G.; Apelt, F.; Carillo, P.; Krause, U.; Feil, R.; Mengin, V.; Lauxmann, M.A.; Köhl, K.; Nikoloski, Z.; Stitt, M.; et al. Getting Back to Nature: A Reality Check for Experiments in Controlled Environments. *J. Exp. Bot.* **2017**, *68*, 4463–4477. [CrossRef] [PubMed]
24. Weimers, K.; Bergstrand, K.-J.; Hultberg, M.; Asp, H. Liquid Anaerobic Digestate as Sole Nutrient Source in Soilless Horticulture or Spiked with Mineral Nutrients for Improved Plant Growth. *Front. Plant Sci.* **2022**, *13*, 770179. [CrossRef] [PubMed]
25. Ferraz-Almeida, R. Balance of Nitrate and Ammonium in Tropical Soil Conditions: Soil Factors Analyzed by Machine Learning. *Nitrogen* **2024**, *5*, 732–745. [CrossRef]
26. Ahmed, R.; Mao, L.; Li, Y.; Ding, J.; Lin, W.; Ahmed, S.; Abbas, A.; Ahmed, W. Effect of Different Fertilizations on the Plant-Available Nitrogen in Soil Profile (0–100 cm): A Study on Chinese Cabbage. *Front. Plant Sci.* **2022**, *13*, 863760. [CrossRef] [PubMed]
27. Scăeţeanu, G.V.; Madjar, R.M. The Control of Nitrogen in Farmlands for Sustainability in Agriculture. *Sustainability* **2025**, *17*, 5619. [CrossRef]
28. Scherer, H.W. Dynamics and Availability of the Non-Exchangeable NH_4-N—A Review. *Eur. J. Agron.* **1993**, *2*, 149–160. [CrossRef]
29. Scherer, H.W.; Feils, E.; Beuters, P. Ammonium Fixation and Release by Clay Minerals as Influenced by Potassium. *Plant Soil Environ.* **2014**, *60*, 325–331. [CrossRef]
30. Sahrawat, K.L. Factors Affecting Nitrification in Soils. *Commun. Soil Sci. Plant Anal.* **2008**, *39*, 1436–1446. [CrossRef]
31. Barłóg, P.; Hlisnikovský, L.; Kunzová, E. Effect of Digestate on Soil Organic Carbon and Plant-Available Nutrient Content Compared to Cattle Slurry and Mineral Fertilization. *Agronomy* **2020**, *10*, 379. [CrossRef]
32. Andersson, H.; Bergström, L.; Djodjic, F.; Ulén, B.; Kirchmann, H. Topsoil and Subsoil Properties Influence Phosphorus Leaching from Four Agricultural Soils. *J. Environ. Qual.* **2013**, *42*, 455–463. [CrossRef] [PubMed]
33. Campos, J.L.; Crutchik, D.; Franchi, Ó.; Pavissich, J.P.; Belmonte, M.; Pedrouso, A.; Mosquera-Corral, A.; Val del Río, Á. Nitrogen and Phosphorus Recovery from Anaerobically Pretreated Agro-Food Wastes: A Review. *Front. Sustain. Food Syst.* **2019**, *2*, 91. [CrossRef]
34. Aronsson, H.; Liu, J.; Ekre, E.; Torstensson, G.; Salomon, E. Effects of Pig and Dairy Slurry Application on N and P Leaching from Crop Rotations with Spring Cereals and Forage Leys. *Nutr. Cycl. Agroecosystems* **2014**, *98*, 281–293. [CrossRef]
35. Mazzini, S.; Borgonovo, G.; Scaglioni, L.; Bedussi, F.; D'Imporzano, G.; Tambone, F.; Adani, F. Phosphorus speciation during anaerobic digestion and subsequent solid/liquid separation. *Sci. Total Environ.* **2020**, *734*, 139284. [CrossRef] [PubMed]
36. Manono, B.O.; Kimiti, J.M.; Musyoka, D.K. Landscape Determinants of Nitrogen Leaching Risk: Mechanisms, Impacts, and Mitigation Strategies. *Nitrogen* **2026**, *7*, 20. [CrossRef]
37. Jarvis, N.J. A Review of Non-Equilibrium Water Flow and Solute Transport in Soil Macropores: Principles, Controlling Factors and Consequences for Water Quality. *Eur. J. Soil Sci.* **2007**, *58*, 523–546. [CrossRef]
38. Hina, N.S. Global Meta-Analysis of Nitrate Leaching Vulnerability in Synthetic and Organic Fertilizers over the Past Four Decades. *Water* **2024**, *16*, 457. [CrossRef]
39. Cugnon, T.; De Toffoli, M.; Mahillon, J.; Lambert, R. Improving Nitrogen Fertilization Recommendations in Temperate Agricultural Systems: A Study on Walloon Soils Using Anaerobic Incubation and POxC. *Nitrogen* **2025**, *6*, 91. [CrossRef]
40. Asomaning, S.K. Processes and Factors Affecting Phosphorus Sorption in Soils. In *Sorption in 2020s*; Kyzas, G.Z., Lazaridis, N., Eds.; IntechOpen: Rijeka, Croatia, 2020. [CrossRef]

Disclaimer/Publisher's Note: The statements, opinions and data contained in all publications are solely those of the individual author(s) and contributor(s) and not of MDPI and/or the editor(s). MDPI and/or the editor(s) disclaim responsibility for any injury to people or property resulting from any ideas, methods, instructions or products referred to in the content.