

Review

Nutrient Recovery Strategies and Agronomic Performance in Circular Farming: A Comprehensive Review

Vaibhav Shrivastava ^{1,*}  and Ikhlas Laasri ²¹ Department of Energy and Technology, Swedish University of Agricultural Sciences, Lennart Hjelms väg 9, P.O. Box 7032, 750 07 Uppsala, Sweden² BETA Tech Center (TECNIO Network), University of Vic-Central University of Catalonia, Ctra. De Roda 70, 08500 Vic, Spain; ikhlas.laasri@uvic.cat

* Correspondence: vaibhav.shrivastava@slu.se; Tel.: +46-765037518

Abstract

Circular agriculture reclaims nutrients from waste streams to reduce fertilizer imports, mitigate environmental impacts, and close material loops. This review evaluates the agronomic performance of nitrogen, phosphorus, and potassium products recovered from wastewater, crop residues, and manure compared with conventional fertilizers. A structured literature survey identified 85 pot and field trials published between 2010 and 2024, covering ammonium salts, struvite, ashes, compost, digestate, biochar, hydrochar, and biostimulants. Ammonium sulfate and nitrate consistently matched synthetic yields (95–105%) due to their solubility and immediate N availability, while aqueous ammonia showed variable results depending on application timing and soil pH. Struvite and phosphorus-rich ashes performed best (90–100%) in neutral to slightly acidic soils, whereas organo-mineral phosphate fertilizers (85–95%) were less effective in alkaline soils. Potassium-rich ashes and waste mica were effective (80–95%) in soils with moderate cation exchange, though mica underperformed (60–75%) in coarse soils. Biochars and hydrochars improved soil water retention and nutrient exchange, yielding 90–110% of synthetic performance, while biostimulants increased yields by 8–20%. Recovered products demonstrate agronomic equivalence while offering co-benefits for soil health, waste management, and circular economy goals. Future work should prioritize long-term field validation, techno-economic analysis, and regulatory integration to enable large-scale adoption.

Keywords: bio-based fertilizers; manure; wastewater; ammonium salts; struvite; biostimulants

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1. Introduction

Global food demand is set to climb sharply as the world population is predicted to approach 10 billion by 2050, placing unprecedented pressure on agricultural systems [1,2]. Yet nutrient availability is uneven: a handful of countries control most phosphate rock and potash reserves, while many others—particularly in Europe—rely heavily on imports to sustain their soils (Table 1). In 2024 alone, the European Union imported approximately 45%, 46%, and 58% of its inorganic nitrogen, phosphates, and potash consumption, respectively [3]. The production of nitrogen, vital for agriculture, entails an energy-intensive process, the Haber–Bosch process, which not only consumes significant energy but also has notable environmental implications [4]. Similarly, countries engaged in large-scale extraction of phosphorus and potash face similar challenges, exacerbating environmental concerns [5].

However, the global supply of these nutrients is increasingly constrained, with reserves concentrated in only a few regions, leading to high import dependency and exposure to price volatility [4]. At the same time, their extraction and processing contribute to serious environmental burdens, including eutrophication, greenhouse gas emissions, and soil degradation [4]. This dual challenge of scarcity and environmental cost underscores the urgency of exploring alternative nutrient sources.

Circular agriculture provides a systemic response to these challenges by closing nutrient loops, reducing dependency on finite resources, and integrating waste valorization into farming systems. Unlike linear “take–make–dispose” models, circular agriculture emphasizes resource efficiency, soil health, and reduced environmental externalities [5]. Recovering nutrients from waste streams thus directly contributes to the Sustainable Development Goals, the EU Circular Economy Action Plan, and climate mitigation targets by reducing fertilizer-related greenhouse gas emissions [5].

Given the energy intensity and emissions associated with conventional nitrogen production, recycling nitrogen from manure and wastewater into fertilizers offers a promising circular solution that reduces reliance on industrial synthesis [3]. More broadly, waste-derived fertilizers enable nutrient recovery from existing stocks, helping to lower environmental impacts while improving regional self-sufficiency [6].

In light of these issues, addressing the geographical disparity of nutrients while minimizing environmental impact presents a pressing challenge. A pragmatic solution lies in the adoption of waste-derived fertilizers, commonly referred to as bio-based fertilizers, sourced from various streams such as crop residues, wastewater, and manure [3]. For instance, Europe alone produces significant quantities of these residues (Table 1), underscoring the vast feedstock potential for circular nutrient recovery.

In pursuit of advancing the objective of nutrient recovery from waste, the European Commission has introduced the concept of RENURE (REcovered Nitrogen from manURE). This initiative extends beyond the confines established by the Nitrous Directive, particularly surpassing the cap of 170 kg total N/ha/yr [6]. Concurrently, the European Commission has enacted the Fertilizer Product Regulation (FPR) [7], which mandates that products derived from manure must adhere to maximum limit values stipulated in the FPR to be recognized as safe fertilizing products [7]. Together, these policies demonstrate both the urgency and institutional momentum for integrating recovered fertilizers into mainstream agricultural practice. Yet their implementation has only recently commenced, resulting in a notable lack of understanding among stakeholders and technology developers regarding the intricacies of the newly enforced FPR [7].

Table 1. Global context for nutrient use, reserves, and waste streams.

Metric	Value	Source
World population (2023)	8.0 billion	[1]
Projected world population (2050)	9.7 billion	[1]
Annual synthetic N fertilizer applied (global)	~110 million t N	[8]
Phosphorus reserve lifetime	~70 years	[9]
Potassium reserve lifetime	~300 years	[10]
Regional share of N fertilizer use	Asia: 56%; Americas: 22%; Europe: 15%; Africa and Oceania: 7%	[8]
Annual manure generated in EU	1.2–1.8 billion t	[11]
Annual crop residues in EU	430 million t	[12]
Annual municipal wastewater generated	~380 km ³	[13]
Annual sewage sludge production (dry solids)	~45 million t	[14]

The objective of this study is to comprehensively evaluate the agronomic performance of nutrients like NPK recovered from diverse waste streams such as wastewater, crop residues, and manure, ensuring that all products assessed originate from nutrient recovery processes. The analysis is based on evidence from both field and pot trials, providing an integrated perspective across scales. A key novelty of this review lies in the inclusion of biostimulants, an emerging class of bio-based products that are increasingly derived from nutrient recovery pathways but have rarely been evaluated alongside fertilizers in terms of agronomic effectiveness. By considering biostimulants together with recovered fertilizers, this review captures a broader spectrum of bio-based inputs relevant to circular agriculture. Furthermore, by aligning recovered nutrient products with their respective Product Function Categories (PFCs) and Component Material Categories (CMCs) under the EU Fertilising Products Regulation (FPR 2019/1009), this review uniquely addresses regulatory uncertainties and provides a policy-relevant classification framework. This approach provides clarity on the regulatory status of bio-based fertilizers, bridging the gap between research, market applications, and policy frameworks. The existing literature has generally evaluated recovered nutrients individually or focused on specific products such as struvite, without offering an integrated assessment across multiple nutrients, product classes, and regulatory perspectives. By bridging agronomic, regulatory, and circular economy dimensions, this study contributes novel insights into how nutrient recovery can accelerate the transition towards sustainable and circular farming systems.

2. Methodology

2.1. Data and Literature Sources

The review's study selection process involved assessing the relevance of studies sourced from various electronic databases, including Web of Science, Journal Storage (JSTOR), ScienceDirect, and Google Scholar. The research strategy employed headings like "biobased fertilizers" and "manure-derived fertilizers", along with corresponding keywords. To refine results, additional filters such as (1) fertilizers, (2) field/pots, (3) manure, and (4) wastewater were applied. These filters were utilized in the "Advanced search" fields of all databases, filtering articles that incorporated these terms in their abstract, title, or text. Furthermore, a manual search supplemented the dataset with additional records, ensuring the inclusion of pertinent studies.

2.2. Study Selection

The selection of the articles for the further assessment was firstly proceeded via elimination of the duplicates. Following that, the remaining articles were entered in Rayyan, a web application to help in the blinded screening. The first assessment of the selected articles was based on the eligibility of titles and abstracts alone. This further led to the categorization of articles as "included" or "excluded". The following criteria were chosen to find out the appropriate studies for a review.

- i. The study had to refer to at least one product derived from waste.
- ii. The study must report the yield/biomass of the tested product.
- iii. The publication year of the study should be between 2010 and 2024.
- iv. If the language of study is English, Dutch, French, German, or Spanish, it should be included.

Scientific articles that did not meet these inclusion criteria were excluded from this study. The exclusion criteria used in this study were as follows.

- i. Studies only considering technological aspects for producing bio-based product were not considered.

- ii. Studies only considering only primary treatments of waste streams (separation, digestate) were not considered.
- iii. Studies reporting heavy metal uptake using waste products were not considered.
- iv. Studies involving microorganisms for improving of fertilizer performance were not considered.
- v. Year of publication: any study carried out before 2014 was not included.
- vi. Language of study: papers available only in Japanese and Chinese languages were excluded.
- vii. Type of publication: review studies and book chapters were not considered.
- viii. Studies with an irrelevant abstract or with no full text available through any source were discarded.

2.3. Data Extraction

For the extraction of data from the shortlisted studies, a Microsoft Excel-based electronic data sheet file was used for compiling the results (Figure 1). The data extracted are as follows: (1) form of bio-based product, (2) technology implemented, (3) scale of implementation—pot trial, pilot trial, or field trial, (4) type of plant harvested, (5) sample size of test, (6) type of waste used, (7) yield/biomass, (8) type of nutrient used, (9) duration of study, and (10) reference—authors' names and DOI of study.

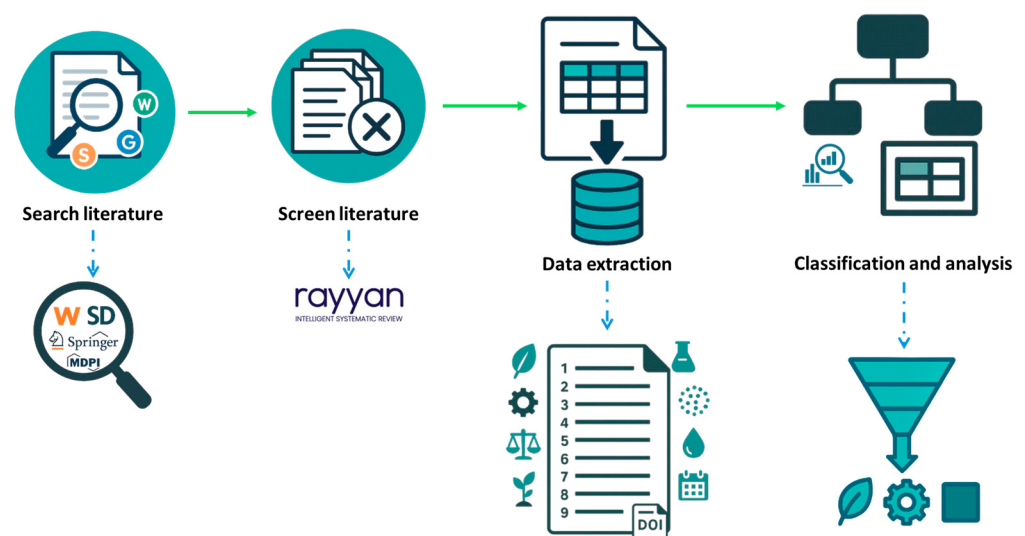


Figure 1. Workflow of review, from literature search and screening in Rayyan to data extraction and EU FPR 2019/1009 classification of recovered nutrient products.

2.4. Overview of Data Collection

In total, we selected 78 studies published between 2010 and 2024. Before any screening was performed, the raw database consisted of 672 data points. Out of these, 183 studies were identified as duplicate sources among the various databases. Further screening (Figure 1) led to a final database consisting of 78 studies reporting plant yield from different waste-derived products.

From among the selected studies, sorting and filtering of the data were carried out in 3 phases.

- In the initial phase, data ($n = 86$) were sorted according to the nutrient recovered from a particular study.
- Phase 2 of the literature survey involved categorizing the types of technology used for nutrient recovery reported in the studies.

- In the 3rd phase of scrutinization, the focus shifted to the products recovered from each technology and their agronomic performance in field and pot trials (Table 2).

Table 2. Treatment process diversification in the selected dataset.

Nutrient Recovered	Product Obtained	Number of Studies
Nitrogen (N)	Ammonium sulfate	15
	Ammonium nitrate	7
	Ammonia water	3
Potassium (K)	Potassium concentrate	3
	Waste mica	3
	Organo-mineral phosphate fertilizer	8
Phosphorous (P)	Phosphorous rich ash	3
	Hydrochar	7
	Biochar	19
	Struvite	18
Others	Biostimulants	13
Total		86 *

* Some studies mention more than one BBF, which is why the number is lower than the total number of individual studies.

2.5. Review Limitations

This review compiles data from various sources including high and low-impact journal articles, non-commercial trial registry records, grey literature, company-owned trial registry records, conference abstracts, and more. The broad range of data sources makes it challenging to verify the quality and authenticity of the information (Figure 2).

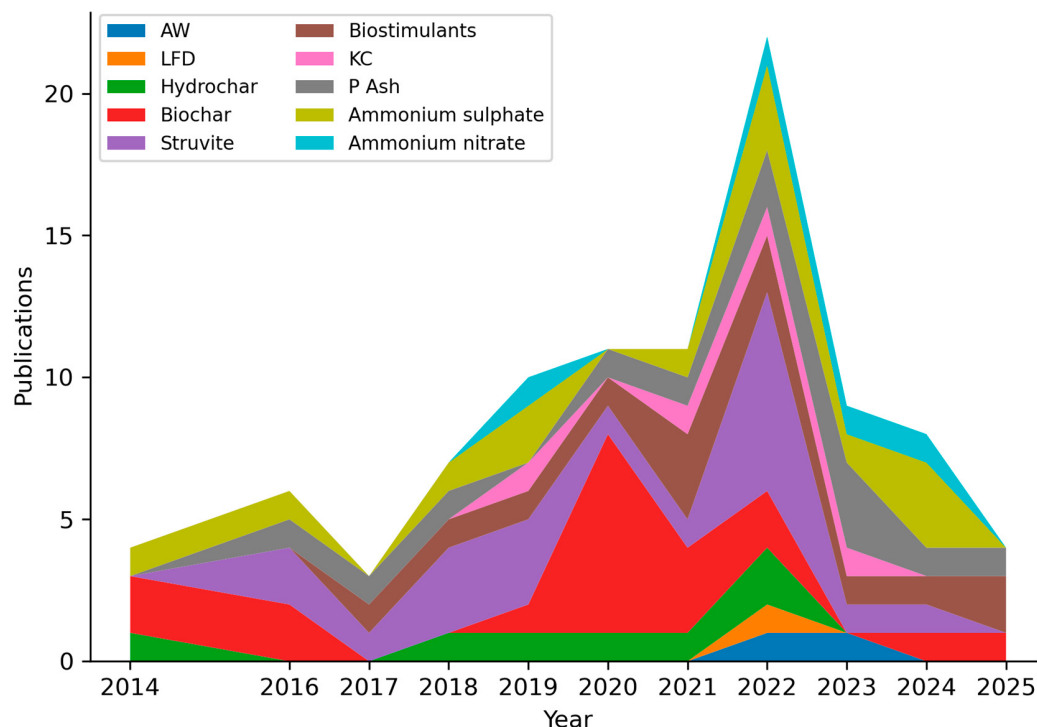


Figure 2. Trends in publications (2014–2025) on waste-derived fertilizers, biostimulants, and soil amendments.

Analysis of the selected dataset reveals that some of the bio-based fertilizers are either still awaiting full-scale implementation or have only been tested for 1–2 years. Additionally,

this review includes some products, both at laboratory and full-scale, currently being used by various businesses to promote products derived from waste streams. Hence, it is important to note that the effectiveness of the products discussed in the studies may be uncertain.

2.6. Data Analysis

The final dataset underwent analysis to identify trends in nutrient recovery technology and their performance in agronomic trials. A comprehensive overview is presented in the following.

3. Results and Discussion

3.1. Ammonium-Based Salts for N Recovery

The process for producing ammonium-based salts utilizes the stripping–scrubbing method. This technique, common in chemical engineering, effectively separates gases or vapors from a liquid solution. NH_3 stripping involves converting aqueous ammonium (NH_4^+) into gaseous NH_3 , primarily achieved through pH and temperature adjustments. The resulting ammonia gas is then introduced into another solution, typically an acidic solution like sulfuric acid (H_2SO_4) or nitric acid (HNO_3). During the scrubbing phase, the ammonia gas reacts with the acid, yielding products such as ammonium sulfate $[(\text{NH}_4)_2\text{SO}_4]$ and/or ammonium nitrate (NH_4NO_3), depending on the specific conditions and reagents used [15,16].

Evaporation is an additional technique that transforms the water-soluble NH_4^+ into gaseous NH_3 , which is then retrieved through condensation as ammonia water (AW). Specifically, the liquid component of the digestate is directed to the evaporator, where NH_3 condensate is collected. Subsequently, the collected evaporation condensate undergoes a process in an NH_3 stripper, yielding AW as the final product. This resulting AW is devoid of suspended particles, metals, or pathogens since it is generated through condensation following evaporation [17].

3.1.1. Ammonium Sulfate (AS)

Regarding the utilization of recovered ammonium sulfate for vegetative growth, numerous studies have investigated its efficacy in both field and greenhouse settings across various crops. For instance, in a study by Rietra et al. [18], ammonium sulfate recovered via animal manure scrubbing was tested in a controlled pot trial for maize and grass. Ammonium sulfate showed similar performance to synthetic CAN (maize: AS 58–62 g/pot, CAN 52–61 g/pot; grass: AS 17–21 g/pot, CAN 21–25 g/pot) for sandy and clayey soils. The study conducted by Horta et al. [19] examined the application of AS recovered from wastewater in a pot experiment involving triticale. The findings revealed a 29% higher yield for ammonium sulfate compared to the commercial Hoagland solution. Similarly, another pot study by Rodrigues et al. [20] assessed the use of wastewater-recovered AS on spinach and radish plants in a slightly acidic soil. The results indicated that the biomass produced for recovered AS (spinach: 30 ± 7.5 g/pot, radish: 35 ± 11 g/pot) was comparable to that obtained with synthetic ammonium sulfate (spinach: 28 ± 4.5 g/pot, radish: 32 ± 7.5 g/pot). Also, Sigurnjak et al.'s [21] study on lettuce also showed a similar performance between manure-recovered AS (254 ± 36 kg/m²) and synthetic fertilizers (244 ± 14 kg/m²) in sandy soil.

The use of waste-derived ammonium sulfate has been extensively explored in field studies over the past decade. In a study by Shrivastava et al. [22], manure-derived AS was applied at 40%, 70%, and 100% of the crop N demand in a four-year maize–spinach–potato rotation. At the 100% rate, yields (ratios 0.86–1.09) closely matched those obtained with

synthetic fertilizers, even under contrasting dry and wet conditions in sandy soil. Similarly, the study by Rietra et al. [18] showed that AS matched CAN yields (maize: AS 10.3 t ha⁻¹, CAN 10.4 t ha⁻¹; grass: AS 8.9 t ha⁻¹, CAN 9.0 t ha⁻¹) under dry conditions in sandy and clayey soils. Additionally, Hendriks et al. [23] investigated its impact on potatoes in sandy soil, while Jin et al. (2022) [24] focused on eggplant using pig manure-derived ammonium sulfate. In both studies, potatoes (AS: 73 ± 9.1 tons/ha, CAN: 68 ± 5.9 tons/ha) and eggplant (AS: 63.42 ± 3.2 tons/ha, Urea: 62.65 ± 3.0 tons/ha) demonstrated performance on par with that of synthetic fertilizers. Similarly, studies on maize have demonstrated comparable yields, falling within the range of 5 ± 11% of synthetic fertilizer yields when using AS recovered from manure sources [15,25–27]. The comparable performance may stem from the readily available nitrogen in the form of NH₄⁺ present in AS [28]. Additionally, in instances where crops thrive in acidic environments for optimal growth, AS offers this benefit due to its relatively low pH range of 2 to 5.

However, in contrast, when it comes to winter plants, the study conducted by Martin et al. [29] on white clovers and ryegrass in temperate grasslands soil, as well as that by Müller et al. [30] on wheat and barley, revealed a slightly lower performance, ranging from 10% to 20% compared to synthetic counterparts. However, in the following year, trailing shoe application of LAS on maize and rapeseed matched synthetic CAN, achieving 51.2 tons/ha and 4.4 tons/ha, respectively. One plausible explanation could be the lower solubility of AS in soil during cold temperatures [31]. This reduced solubility may result in significant leaching of AS into groundwater, thereby diminishing its availability to the crop itself [32].

Additionally, Robles-Aguilar et al. [33] investigated the performance of manure-derived AS blended with synthetic PK on viola plants. The results revealed relatively lower performance (<1 g/pot) compared to synthetic counterparts (>4 g/pot). This disparity could be attributed to higher ammonium accumulation in the treatments with AS. This observation might be elucidated by a decrease in the pH of the growing medium, potentially inhibiting nitrification processes [33]. However, it is important to note that unlike perennial crops, ornamental plants, such as violas, are typically more sensitive to the pH of products used (due to their relatively lower stress tolerance), which could contribute to the lower biomass observed in this case [34].

3.1.2. Ammonium Nitrate (AN)

The evaluation of wastewater-recovered AN was carried out in pot tests across studies involving tomatoes [35,36], spinach, radish [20], and viola [33] in sandy soils. In these studies, AN demonstrated comparable performance to synthetic alternatives, with values ranging from 10 ± 15% to the synthetic fertilizer yields. Additionally, AN derived from pig manure [17] also exhibited similar efficiency, with AN (53 ± 7.8 g/pot) performing comparably to synthetic fertilizer (56 ± 6.1 g/pot) for lettuce. Additionally, in a similar study on lettuce conducted by Sigurnjak et al. [15] using two different soils, manure-derived AN (sandy soil: 89 ± 8 g/pot; loamy soil: 76 ± 4 g/pot) demonstrated a higher yield than synthetic fertilizer (sandy soil: 77 ± 2 g/pot; loamy soil: 69 ± 2 g/pot). The higher yield could be attributed to the optimal growth conditions for lettuce plants, which require acidic soil with a pH of 6.0–6.5. This acidic environment may contribute to better yields compared to CAN, which has a relatively neutral pH [37].

Limited studies have been conducted in the last decade regarding the testing of recovered AN on a field scale. In a study by Shrivastava et al. [22], manure-derived AN applied at 40%, 70%, and 100% of crop N demand produced yield ratios of 0.49–1.02 at the 100% rate over a four-year maize–spinach–potato rotation, indicating somewhat variable performance relative to the synthetic fertilizer. This variability likely reflects

differences in N release dynamics and soil interactions of manure-derived AN [22]. In a study by Sigurnjak et al. [15], maize grown on sandy soils under wet conditions showed no significant differences in yield between pig manure-recovered AN (59 ± 6 t/ha) and synthetic fertilizer (57 ± 6 t/ha). Similarly, in the study by Saju et al. [25], despite the maize field trial being affected by a combination of dry and wet weather conditions, AN showed a yield of 32 ± 9.9 tons/ha compared to the synthetic fertilizer yield of 36 ± 7.5 tons/ha.

The reason behind the consistent performance observed in the aforementioned studies using AN is similar to AS. This is primarily attributed to the presence of readily available plant nitrogen (NO_3 and NH_4^+), which effectively supports plant growth [15,25]. Moreover, the liquid nature of AN offers advantages for precision fertilization [38]. However, inherent risks include volatility and leaching during application [39]. Additionally, the high application rates of liquid AN may lead to disturbances in soil pH compared to synthetic fertilizers, due to its liquid form and lower total N content [40].

3.1.3. Ammonia Water (AW)

Ammonia water, also known as ammonium hydroxide (NH_4OH), is a product commonly used in de- NO_x applications in various industrial plants [41]. However, its potential as a bio-based fertilizer remains largely unexplored across studies to date. In total, only three studies exist discussing ammonia water as a BBF. In a study by Saju et al. [25], AW derived from pig manure was evaluated in lettuce pot cultivation in sandy soils, with the efficiency of AW (53 ± 7.8 g/pot) found to be equivalent to that of synthetic fertilizer (56 ± 6.1 g/pot). In another study by Shrivastava et al. [42], AW was tested at two different pH levels (acidified pH 5 and initial pH 11) to examine the effect on lettuce biomass in sandy soils. The study revealed that non-acidified AW demonstrated higher biomass (71.5 ± 3.0 g/pot) compared to acidified AW (62.7 ± 15.8 g/pot), which in turn performed similarly to synthetic fertilizer (72.72 ± 2.17 g/pot).

Additionally, Luo et al. [43] demonstrated the performance of AW in a field trial experiment with maize as the test crop. The experiment revealed similar overall performance of AW (13.3 ± 2.1 tons/ha) compared to synthetic fertilizer (15.1 ± 1.1 tons/ha). However, when comparing the total root depth, AW resulted in significantly lower performance (3.5 ± 2.0 m) compared to synthetic fertilizers (11.7 ± 4.5 m). The shallower root development can be attributed to the inhibition of rhizospheric bacteria due to the increase in soil pH [44]. AW, with its high pH (~ 11), contrasts with the pH range of 5.8 to 6.0 typically required for proper maize growth [45]. Additionally, the study revealed higher soil synthetic nitrogen levels after harvest in plots treated with AW, indicating the potential for increased leaching [43]. This dual effect of elevated soil pH and potential nitrogen leaching may further contribute to inadequate root development in maize.

Even though AW has demonstrated equal performance to synthetic fertilizers across various studies, the limited research in this field may be attributed to farmers' reluctance to use it as a fertilizer. Firstly, the high pH of ammonia water makes it highly volatile, resulting in substantial ammonia emissions into the atmosphere during application [42]. Additionally, the general boiling point of 30% AW is approximately 37°C [46]. This low boiling point makes it unsuitable for field application, especially in high-temperature regions, as much of the solution would evaporate before reaching the soil, diminishing its effectiveness as a fertilizer [42]. Secondly, the strong smell of ammonia itself poses a potential issue, as it may be unpleasant and create a nuisance during application [43].

Across crops and soils, recovered ammonium salts' (AS, AN, AW) effectiveness is enhanced in acidic or sandy soils where ammonium uptake is favored, but performance can decline in colder or high-pH environments due to reduced solubility, leaching, or volatilization risks. From a practical standpoint, these findings suggest that recovered AS and AN

are suitable substitutes for conventional N sources in most temperate systems, while AW use remains constrained by its instability and volatilization losses. Strategically, recovered ammonium salts can lower dependence on energy-intensive Haber–Bosch N fertilizers, provided site-specific soil pH, climate, and crop requirements are carefully managed.

3.2. Potassium-Based Extracts

Evaporator concentrate (KC) is a by-product obtained from the stripping–scrubbing process. It is typically rich in potassium and contains other non-volatile nutrients that remain after the evaporation step. During this process, most of the nitrogen is removed, leaving behind a concentrated product known as KC [23]. The obtained KC can be modified to retain ammonia via acidification (with H_2SO_4) prior to the evaporator. This modification leads to KC with a high $\text{NH}_4\text{-N}$ /total N ratio. However, a significant amount of sulfur is also introduced during the acidification process [23].

In addition to the evaporation method, there are other methods available for extracting potassium from waste. Mineral and organic acids are known to be efficient in extracting potassium from rocks and minerals, making them beneficial for extraction from waste materials [47]. Common methods include water extraction, neutral ammonium acetate extraction, citric acid extraction, and boiling nitric acid extraction.

3.2.1. Potassium Concentrate (KC)

The KC obtained from pig manure has demonstrated variable performance compared to synthetic fertilizers in pot experiments assessing agronomic efficiency. In a study by Hendriks et al. [23] involving potato cultivated in sandy soils, KC exhibited higher efficiency (0.4 ± 0.02 kg/pot) compared to synthetic potassium fertilizer (0.3 ± 0.05 kg/pot). However, maize pot trials by Luo et al. [47] showed lower yield and root biomass for KC (2.7 ± 0.2 g/pot; 2.4 ± 0.1 g/pot) compared to synthetic fertilizer (3.8 ± 0.6 g/pot; 3.0 ± 1.0 g/pot). Even though the pot trial conducted by [47] primarily focused on comparing the performance of N fertilizers, it is worth noting that the K uptake in the plants was comparatively lower for KC (157 ± 16 mg/plant) compared to the synthetic reference (233 ± 43 mg/pot). This outcome could be due to the addition of Na resulting from the application of KC. This Na influx may have hindered the initial growth of young plants, consequently impacting the overall yield and K uptake in the plants. However, in the case of field trials, this effect was not observed for maize, as KC performed similarly (42 ± 3 tons/ha) to synthetic N fertilizer (45 ± 5 tons/ha) under wet weather conditions [47].

3.2.2. Waste Mica (WM)

The pot trials conducted on waste mica (WM) consistently showed lower efficiency compared to synthetic K fertilizers. In a study by Basak [48] on K-deficient Alfisol soil, WM was tested on palmarosa (68.89 g/pot) and sudangrass (47.93 g/pot), resulting in lower efficiency compared to synthetic fertilizer (72.23 g/pot; 51.08 g/pot). This is primarily attributed to the smaller particle size of WM, which enhances biomass efficiency and K uptake. The larger surface area available for biogeochemical reactions in the rhizosphere allows for more potassium release from finer particles [48]. Similarly, in a study by Biswas et al. [49], WM was tested on wheat in Alfisol, with double the dosage compared to synthetic fertilizer. However, the yield was only $\sim 25\%$ (7 ± 3 g/pot) compared to synthetic potassium fertilizers (24 ± 2 g/pot). This can be attributed to WM being a slow-release fertilizer, which stabilizes K in the soil more effectively than synthetic K fertilizer [49]. Notably, in the same study, the soil K balance after harvest was negative with synthetic fertilizer, whereas a positive trend was observed with WM [49].

In contrast to the pot trials, in field trials conducted by Pramanik et al. [50] on tea as a test crop under a subtropical climate, WM showed similar efficiency to synthetic fertilizers

over a period of two years. This can be attributed to the pH of the soil, as tea plants prefer acidic soil with a pH of around 4.0 to 5.5 [51], which can be induced using waste mica. The secretion of root exudates in the rhizosphere induced by waste mica alters the rhizosphere pH by releasing H^+ ions [48]. This has important implications for mineral dissolution, thereby increasing the efficiency of WM.

Overall, both KC and WM show mixed results compared to synthetic K fertilizers, with performance strongly moderated by soil type and crop physiology. KC appears more effective in sandy soils and under field conditions, but sodium release can reduce plant K uptake in sensitive crops. WM, by contrast, often underperforms in short pot trials due to its slow-release nature, yet field evidence from acidic soils (e.g., tea plantations) suggests that it can be agronomically viable where gradual K supply is advantageous. Practically, this indicates that KC may serve better in intensive systems with immediate K demand, while WM has potential in perennial or acid-savoring crops where long-term soil fertility and pH adjustment are beneficial.

3.3. Organo-Mineral Phosphate Fertilizers (OMFs)

Organo-mineral phosphate fertilizer is a blend of organic medium with synthetic P fertilizer and essential nutrients [52]. It can be created using organic amendments such as various types of manure (e.g., chicken, poultry, pig, cow) or plant residue-based materials (sugarcane, sawdust, etc.) rich in phosphorus and other macro/micro nutrients [52]. Additionally, different fractions of P fertilizer can be mixed depending on the desired composition.

The preparation of OMF typically involves three main steps (Figure 3). Firstly, both the organic medium and synthetic fertilizer, such as triple superphosphate (TSP) or monoammonium phosphate (MAP), are oven-dried at around 60 °C until a constant weight is achieved [52,53]. Next, they are ground separately and mixed in an industrial mixer before being sieved to achieve a desired particle size, typically 3 to 5 mm. After determining the required total P content, the mixture is further blended in an industrial mixer with a secondary catalyst like bentonite and sodium silicate [52,53]. Necessary micro and macro nutrients are then added based on the specific crop requirements for the particular OMF blend. The mixture is then transferred into a disk granulator to form granules, which are dried again at 60 °C until a constant weight is reached [52,53]. The resulting granules typically have a size between 3 and 4 mm [52,53].

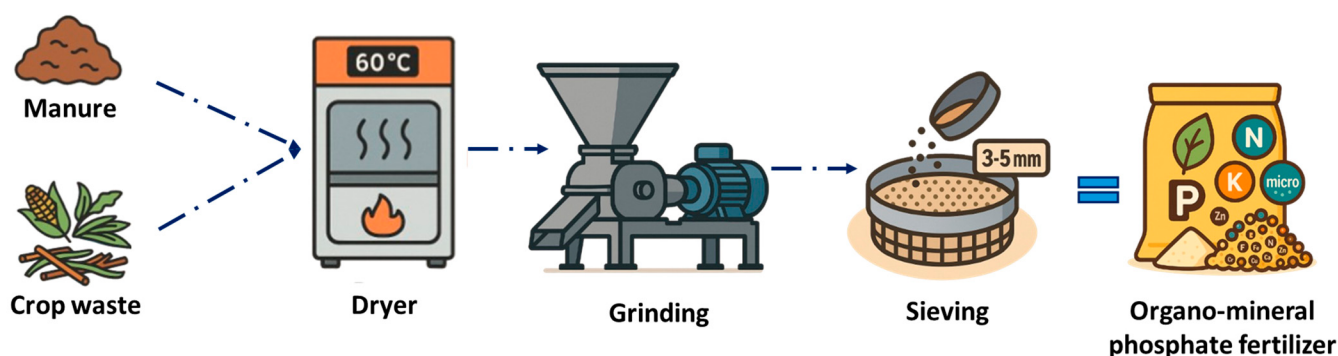


Figure 3. A process diagram illustrating the production of organo-mineral fertilizers (OMFs) from manure and crop residues.

3.3.1. Manure-Derived OMF

Several studies have investigated the effect of OMF derived from chicken manure on maize growth. Sá et al. [54] conducted pot experiments and found that the biomass of maize was similar between OMF (58 ± 6 g/pot) and monoammonium phosphate

(MAP) (49 ± 6 g/pot) across four different experiments. Similarly, Fasaro et al. [55] conducted a study on maize, demonstrating that OMF (64 ± 7 g/pot) performed similarly to triple superphosphate (TSP) (59 ± 6 g/pot) in oxisol soils at a P dose of 100 kg/ha. Additionally, Sakurada et al. [56] and Nascimento et al. [57] also observed a comparable trend in yield, with OMF (8 ± 1 g/pot; 13 ± 4 g/pot) performing similarly to synthetic fertilizers (6 ± 1 g/pot; 11 ± 1 g/pot). However, [55] conducted experiments not only in oxisol soils but also in Entisol soil. In the case of Entisol soil, the performance of OMF (59 ± 7 g/pot) was lower compared to TSP (75 ± 6 g/pot).

This difference in performance between OMF and TSP in two different types of soil can be primarily attributed to the maximum phosphorus absorption capacity. In Entisol soil, TSP's higher phosphorus solubility led to better plant growth due to increased soil phosphorus availability compared to OMF [55]. Conversely, in oxisol soil, which has a high phosphorus absorption capacity, OMF's higher solubility resulted in similar yields to TSP [55]. This highlights the importance of considering soil characteristics and phosphorus availability when selecting fertilizers for optimal plant growth.

3.3.2. Crop Waste-Derived OMF

Various studies have investigated the use of vegetable waste as OMF, assessing crop efficiency in pot trials. In a study by Sitzmann et al. [58], OMFs based on green compost, municipal solid waste compost, and vermicompost were tested on tomato, yielding a shoot + fruit biomass of 47.3–65.8 g/pot vs. 58.2 g/pot under synthetic NPK fertilization. Similarly, Vieira et al. [59] used sugarcane waste cake as OMF on cabbage and spinach. For cabbage, the efficiency was found to be similar to that of synthetic fertilizer (308 g/pot; 281 g/pot), whereas for lettuce, the yield was reduced (19 g/pot; 45 g/pot). The lower yield observed in the case of spinach can be attributed to the slow-release action of the OMF, resulting in reduced availability of phosphorus for the crop. Spinach typically requires phosphorus to be quickly taken up [60], and the slower release of phosphorus from OMF may not meet the crop's demand during its critical growth stages, leading to lower yields. In contrast, cabbage may tolerate the slower release of P better compared to spinach [59], resulting in similar yields between OMF and synthetic fertilizer. In another study by Erenoğlu et al. [61], where wheat was the test crop, similar efficiency was observed between OMF (3.0 ± 0.3 g/pot) and synthetic fertilizer (2.9 ± 0.2 g/pot). Furthermore, Magela et al. [62] tested OMF derived from composted biowaste and found that it produced tuber yields comparable to synthetic fertilizer for two potato cultivars (Ágata: OMF 37.8 tons/ha vs. control 39.5 tons/ha; Atlantic: OMF 50.7 tons/ha vs. control 47.3 tons/ha).

Overall, in soils with a high P fixation capacity, such as Oxisols, OMFs can match TSP or MAP, while in lighter soils (e.g., Entisols), the lower solubility of OMF may limit short-term P availability. These findings highlight the potential of OMFs as a circular alternative to mineral fertilizers, particularly in regions with acidic, highly weathered soils where nutrient retention is critical. From a practical standpoint, OMFs valorize organic residues while supplying nutrients, but their successful adoption will require soil-specific recommendations and blending strategies to ensure reliable crop performance.

3.4. Chars

The process of obtaining chars for nutrient recovery has been extensively discussed in recent years. The recovery of nutrients from chars depends on several factors, including the type of process, temperature, and substrate used [63]. These factors categorize the process into three main sectors: mono-incineration/co-incineration, primarily aimed at obtaining phosphorus, and the production of hydrochar and biochar, aimed at generating a nutrient-rich substrate that also serves as an organic amendment [63].

For incineration, various product streams can be utilized, such as sewage sludge, tomato waste leaves, or other non-edible plant residues, to obtain P ash [64]. To obtain this, the substrate is incinerated in a fluidized bed combustor at temperatures ranging between 850 °C and 1000 °C, resulting in ash formation [64]. Further processing of the ash to recover P involves using different extraction methods (acid extraction, CaCl_2 extraction, etc.); as phosphorous ash may contain a high amount of P, it may not be readily available for plant uptake [65].

For the production of biochar, thermochemical methods such as pyrolysis, gasification, and flash carbonization are commonly used [66]. These methods involve the decomposition of dry feedstock at high temperatures ranging from about 400 °C to 700 °C, in the absence or partial supply of oxygen [66]. The type of yield obtained from this process depends on factors like heating rate and residence time. On the other hand, hydrochars are produced from wet feedstock, as biochar production from such feedstock may not be very suitable and would require more energy [63]. The production of hydrochar involves a process called hydrothermal carbonization, where wet feedstock is subjected to subcritical conditions at temperatures ranging between 180 °C and 250 °C and pressures ranging from 2 MPa to 6 MPa [63].

3.4.1. Phosphorous-Rich Ash

In a study by Cabeza [67], two different types of P ash derived from animal waste and from sewage sludge were tested and applied to maize as a test crop over a two-year pot trial. The experiment indicated that both ashes performed significantly poorer (28 ± 0.6 and 29 ± 1.1 mg P uptake/kg soil) than synthetic fertilizer (42 ± 0.7 mg P uptake/kg soil) in terms of phosphorus uptake. This effect was also observed in yield, with a reduction of 25–30% in the case of ashes compared to TSP. This could be attributed to the absorptive nature of the ashes in the soil, hindering the isotopic exchange of phosphorus in the products and resulting in lower phosphorus uptake by maize [68]. Similar results were obtained in a study by Bogdan et al. [69], where two types of sewage sludge ash obtained from wastewater were tested on ryegrass in a pot experiment in sandy and clayey soils. Both ashes performed significantly lower in terms of P uptake (4.9 ± 0.4 mg/pot and 8.8 ± 1 mg/pot) compared to TSP (13 ± 1 mg/pot). In another study by Dombinov et al. [70], P ash derived from sugarcane husk was tested on soybeans as a test crop in arable soil. Even with a four-time P dosage of ash compared to TSP, P ash performed significantly lower (3.5 ± 0.1 mg/pot) in terms of P uptake compared to TSP (6.0 ± 0.2 mg/pot). A similar trend was also observed for yield, with a 30–40% decrease in dry matter observed for ash compared to TSP.

3.4.2. Hydrochar

Across the studies, hydrochar has generally shown yields comparable to or lower than negative controls, with performance strongly influenced by soil type and growing environment. De Jager et al. [71,72] tested hydrochar from liquid pig manure on Chinese cabbage in greenhouse pot trials across three contrasting European soils—loamy Chernozem, sandy Podzol, and clayey Gleysol—and found no significant yield differences (0.3 ± 0.05 g/pot) compared to the control (0.4 ± 0.1 g/pot), irrespective of particle size. Similarly, Huezio [73] and Yin et al. [74] evaluated manure-derived hydrochar on lettuce under temperate greenhouse conditions, reporting no yield advantage (17 ± 2 g/pot vs. 15 ± 2 g/pot in the control), even when applied to silty loam soils. Reibe et al. [75] observed comparable results for wheat grown in sandy loam under controlled conditions, again showing no significant improvement with hydrochar (4.54 ± 1.8 g/pot vs. 6.69 ± 2.3 g/pot in the control). In contrast, Melo et al. [76,77] conducted trials on beans in highly weathered

tropical Oxisols in Brazil and recorded 20–30% higher dry matter yields with hydrochar, although P and K uptake remained unchanged. The major reason for the lack of performance of hydrochar in pot trials for different crops is likely due to the absence of plant available nutrients in the hydrochar [76]. Collectively, these studies suggest that while hydrochar contributes little as a direct nutrient source in temperate greenhouse systems, its performance may improve under tropical, nutrient-poor soils, aligning with its primary role as a soil amendment to enhance structure, water retention, and carbon sequestration rather than as a direct P or K fertilizer [78,79]. Additionally, the primary use of hydrochar is often for soil amendment purposes and to enhance the natural process of coalification, thereby improving soil structure, water retention, and carbon sequestration, rather than solely as a P or K fertilizer [78]. Furthermore, the phytotoxic effects of hydrochar, particularly those derived from sewage sludge containing elevated concentrations of heavy metals exceeding regulatory limits, could contribute to lower yields [79].

3.4.3. Biochar

The agronomic performance of waste-derived biochar across various crops has been extensively studied over the years. Waste streams commonly used for the production of waste-derived biochar include manure-based streams, crop residue-based streams, animal bone-based streams, and sewage sludge. In a study by Krounbi et al. [80], dairy manure-derived biochar was tested in pot-based experiments using three different crops, radish, marigold, and tomato, planted in a slightly acidic sandy soil. In terms of plant biomass, the biochar performed similarly to synthetic fertilizer for all three crops ($\text{Yield}_{\text{synthetic}}/\text{Yield}_{\text{Biochar}} = 0.9\text{--}1.05$). However, there was a significant difference in N uptake, with the biochar-treated plants showing a 20–30% lower N uptake compared to the synthetic treatments across all three cases. This might be due to the lower N content in the biochar, which could contribute to its reduced availability for plant N uptake. However, in contrast, research by Nardis et al. [81] demonstrated promising results with Mg^{2+} -loaded biochar derived from pig and poultry manure. In pot experiments with maize as the test crop, both biochars ($\text{Biochar}_{\text{poultry}}: 1.0 \pm 0.03 \text{ g/pot}$; $\text{Biochar}_{\text{pig}}: 0.7 \pm 0.05 \text{ g/pot}$) outperformed TSP ($0.65 \pm 0.1 \text{ g/pot}$) in terms of dry matter. Similarly, in experiments by Carneiro et al. [82] with maize and bean plants using poultry litter biochar in Oxisols, higher yields ($\text{Yield}_{\text{synthetic}}/\text{Yield}_{\text{Biochar}} = 1.1\text{--}1.2$) and similar P uptake ($\text{Yield}_{\text{synthetic}}/\text{Yield}_{\text{Biochar}} = 0.8\text{--}1.1$) were observed compared to TSP. This could be attributed to the synergistic effect of Mg on P absorption. Mg serves as an enzymatic activator in phosphorelative enzymes, facilitating the bridging of ATP and/or ADP in the enzyme molecule [81]. This synergistic effect enhances phosphorus absorption, as magnesium activates the ATPase of the membrane responsible for ionic absorption [81].

In a study by Ali et al. [83], biochar derived from cassava straw was evaluated over three crop cycles of noodle rice. The soil treated with biochar exhibited higher grain yield ($116.7 \pm 5.1 \text{ g/pot}$) and dry biomass ($188.7 \pm 8.3 \text{ g/pot}$) compared to the negative control ($96.6 \pm 5.12 \text{ g/pot}$; $177.8 \pm 3 \text{ g/pot}$). Furthermore, the effectiveness of biochar derived from rice husks on wheat cultivation has been investigated in studies by Kulczycki et al. [84] and He et al. [85]. In both studies, soil amended with rice husk biochar exhibited significantly higher performance compared to control conditions ($\text{Yield}_{\text{synthetic}}/\text{Yield}_{\text{Biochar}} = 0.7\text{--}0.9$). Furthermore, Hammerschmiedt et al. [86] used biochar derived from agricultural waste and tested on barley in a pot experiment. The research revealed that soil treated with biochar + manure resulted in a higher biomass ($21 \pm 1.0 \text{ g/pot}$) compared to the negative control treatment ($10 \pm 0.8 \text{ g/pot}$). Additionally, the potential of biochar derived from crop waste for enhancing ryegrass growth has been explored in studies by Manolikaki [87] and Rosa et al. [88]. In pot-based experiments, biochar derived from rice husk, grape

pomace, and olive pruning demonstrated 2–3 times higher performance in terms of biomass production compared to negative control conditions in both studies.

The research on biochar's effect on maize as a test crop has yielded promising results across multiple studies. In Smider and Singh [89], biochar from tomato green waste demonstrated higher efficiency (7.9 ± 0.2 g/pot) than control conditions (2.3 ± 0.1 g/pot) in a seven-week pot experiment. Similarly, biochar derived from brewer spent grain, as studied by Manolikaki [90], increased maize dry yield by 60% compared to the negative control (3.1 ± 0.1 g/pot). Additionally, biochar from grape pomace and rice husks, researched by Manolikaki [91], showed a nearly 50% increase over control conditions in a pot experiment. In another study by Banik [92], biochar from red oak residue combined with manure performed similarly (4.0 ± 0.8 g/pot) to synthetic fertilizer (3.4 ± 0.9 g/pot).

The utilization of other waste streams in experiments has also shown promising results. For instance, in the study by Piri and Sepehr [93], biochar derived from sewage sludge was applied to maize as a test crop, resulting in a significant increase in maize biomass by 30–40% compared to TSP. Similarly, in a study by Fang et al. [94], sewage sludge biochar was tested on two different crops, choy sum and dry grass, where it performed comparably to synthetic fertilizer in terms of crop yield ($\text{Yield}_{\text{synthetic}}/\text{Yield}_{\text{Biochar}} = 0.9\text{--}1.2$). Additionally, biochar derived from animal bones has demonstrated promising results in enhancing crop growth. In the study by Carella et al. [95], maize growth was significantly enhanced when the soil was treated with 10% biochar derived from fish bones, resulting in four times higher dry biomass compared to negative control conditions. Similarly, in a field trial conducted by Panten [96] over a period of five years on barley, oilseed rape, wheat, lupine, and rye, animal bone-based biochar showed comparable efficiency to TSP in terms of grain efficiency and yield ($\text{Yield}_{\text{synthetic}}/\text{Yield}_{\text{Biochar}} = 0.85\text{--}1.1$).

The yield increases observed in biochar-amended systems can be explained by its soil and climate-specific effects. In slightly acidic soils, biochar raises pH towards neutrality, which enhances root development and nutrient uptake, particularly in crops sensitive to low pH stress [97]. In tropical and subtropical regions with highly weathered soils (e.g., Oxisols), biochar improves cation exchange capacity, thereby retaining essential nutrients that would otherwise leach rapidly under heavy rainfall conditions. Conversely, in temperate sandy soils with a low water-holding capacity, biochar's porous structure increases soil moisture retention, enabling crops to withstand periods of drought stress and stabilizing yields [98]. Furthermore, the long-term stability of biochar allows it to act as a nutrient reservoir, reducing leaching losses of N and P—especially when co-applied with organic amendments like manure—providing a more consistent nutrient supply over successive growing seasons [99]. These benefits highlight why biochar is often more effective in degraded, acidic, or coarse-textured soils and in climates where rainfall variability or nutrient leaching constrain crop productivity.

The effect of biochar on carbon sequestration and soil quality has been demonstrated in multiple field studies. In a three-year trial, Li et al. [100] applied apple branch-derived biochar at 24–96 tons/ha to loess soil, resulting in a 25% increase in total organic carbon, boosts in soil aggregate stability, and an average cohesion of 36% (peaking at 93% in year 3). Over an eleven-year period, Gross et al. [101] showed that biochar co-applied with compost at 31.5 Mg/ha in loamy soils increased soil organic carbon stocks by 38 Mg/ha, with these gains remaining stable, whereas sandy soils experienced a gradual decline in sequestration benefits. These findings highlight that feedstock type, soil texture, and amendment combinations underpin biochar's long-term stability and its ability to improve soil structure.

In overall, biochar and hydrochar differ in their agronomic roles. Hydrochar generally shows limited yield effects under temperate greenhouse or loamy soils, but can provide

benefits in nutrient-poor tropical Oxisols, aligning more with its role as a soil conditioner than a nutrient source. In contrast, biochar consistently enhances yields in acidic, sandy, and highly weathered soils by improving pH, nutrient retention, and water-holding capacity, while also delivering long-term carbon sequestration. In practice, this means that hydrochar may be best positioned as a carbon-rich soil amendment with climate benefits, whereas biochar offers both agronomic gains and climate resilience, making it more attractive for adoption in regions facing soil degradation, drought stress, or fertilizer dependency.

3.5. Struvite

Struvite is a mineral compound primarily composed of magnesium, aluminum, and phosphate ($\text{NH}_4\text{MgPO}_4 \cdot 6\text{H}_2\text{O}$), commonly formed in environments rich in these elements, such as wastewater treatment plants or manure processing systems [102]. Struvite production involves precipitating the compound out of a solution with excess nutrients, often due to organic matter decomposition [102]. Chemical additives like magnesium sulfate or chloride can enhance this precipitation process. Factors like pH, temperature, and stirring conditions in the reactor greatly influence struvite production [103]. If not removed, struvite crystals can cause issues like pipe and equipment scaling [104].

3.5.1. Struvite Derived from Wastewater Systems

The agronomic performance of struvite derived from wastewater has been extensively studied over the years. In two greenhouse-based experiments conducted over two years by Carreras-Sempere et al. [35,36], wastewater-derived struvite was evaluated as a fertilizer for tomato crops in a hydroponics system. The results consistently showed that the performance of tomato crops treated with wastewater-derived struvite (range: 20.9–24.2 kg/m²) was comparable with TSP (20–25.3 kg/m²). Additionally, assessments of fruit quality parameters revealed similar outcomes between the struvite-treated (223–267 g/fruit) and TSP-treated crops (248–290 g/fruit) in both studies.

In a study by Talboys et al. [105], wheat and buckwheat were tested as test crops in pot experiments lasting 30 and 90 days in sandy soil. The results showed lower P uptake from struvite compared to DAP after 30 days for wheat ($\text{Uptake}_{\text{struvite}}/\text{Uptake}_{\text{DAP}} = 0.3\text{--}0.35$). However, in the 90-day experiment, P uptake ($\text{Uptake}_{\text{struvite}}/\text{Uptake}_{\text{TSP}} = 0.9\text{--}1.05$) and grain yield ($\text{Yield}_{\text{struvite}}/\text{Yield}_{\text{TSP}} = 0.9\text{--}1.1$) from struvite was similar to that of TSP, suggesting its potential as a slow-release P fertilizer for wheat. Interestingly, buckwheat exhibited similar efficiency in P uptake ($\text{Uptake}_{\text{struvite}}/\text{Uptake}_{\text{TSP}} = 0.8\text{--}0.9$) from struvite even after 30 days, unlike wheat, which required 90 days to achieve comparable efficiency. In another study on wheat by Degryse et al. [106], similar efficiency in terms of dry matter yield ($\text{Yield}_{\text{struvite}}/\text{Yield}_{\text{MAP}} = 0.95$) and uptake ($\text{Uptake}_{\text{struvite}}/\text{Uptake}_{\text{MAP}} = 0.97$) was noted between monoammonium phosphate (MAP) and struvite in silty-loamy soil. However, an interesting observation was made regarding efficiency depending on the structure of the struvite. Two structures were tested: granular and ground struvite. The study found that the efficiency was comparable when using ground struvite, but significantly lower, around 25%, compared to MAP when using granular struvite. This difference may be attributed to the lower solubility of struvite in granular form, which the study reported as just 3% of total P soluble in water. Similarly, a study by Martens et al. [107] investigated the effect of incremental dosages of P on spring wheat and alfalfa grass in a field-based experiment. For spring wheat, dosages ranged from 0 to 40 kg P/ha, while for alfalfa grass, the range was from 0 to 90 kg P/ha. In both cases, the study found a linear increase in grain yield (1.92–2.62 Mg/ha) and P accumulation (6.5–9.5 kg/ha) for wheat and in forage yield (2.74–6.31 Mg/ha) and P uptake (2.99–10.87 kg/ha) for alfalfa grass with increasing P

dosages. This suggests a positive correlation between struvite application rates and crop productivity for both crops.

A study by Cabeza [67] investigated the efficacy of recovered struvite from three different wastewater treatment plants as a P source for maize in a two-year pot experiment in sandy and loamy soils. The study found similar P uptake ($\text{Uptake}_{\text{struvite}}/\text{Uptake}_{\text{TSP}} = 0.95\text{--}1.16$) and yield efficiency ($\text{Yield}_{\text{struvite}}/\text{Yield}_{\text{TSP}} = 0.89\text{--}1.02$) across all three struvite sources, indicating their comparable performance to TSP. In a study by Omidire et al. [108] on wheat tested in the field, struvite was precipitated from two different sources: chemically and electrochemically. The performance was compared against DAP, MAP, and TSP. The P uptake with chemically precipitated struvite (187 kg/ha) was similar to that with TSP (187 kg/ha), MAP (205 kg/ha), and DAP (179 kg/ha). However, electrochemically precipitated struvite exhibited significantly higher P uptake (228 kg/ha) compared to other treatments. Similarly, the electrochemically precipitated struvite (11.5 t/ha) showed higher performance in terms of dry yield in comparison to TSP (10 t/ha), MAP (10.5 t/ha), and DAP (9.2 t/ha). Similarly, Mehta et al. [109] also showed the similar performance of struvite to a synthetic alternative in a maize pot trial. Furthermore, a study by Kokulan et al. [110] demonstrated a higher yield for struvite (23.3 Mg/ha) compared to MAP (20 Mg/ha). Moreover, study by Valle et al. [111] tested struvite alongside a composite of thermoplastic starch (TPS) in different concentrations, comparing its performance with TSP. The study found that both struvite (5.5 ± 0.25 g/pot) and TSP (5 ± 1 g/pot) performed comparably in terms of dry yield. However, the performance of a 1:3 blend of TPS/struvite showed higher performance (5.85 ± 1.25 g/pot). The enhanced performance of the composite is attributed to the combined slow-release behavior of struvite in acidic medium and the physical barrier provided by TPS against fast solubilization [111]. This combination makes it suitable for acidic environments like the plant root region, resulting in better performance.

Regarding the performance of struvite on fodder crops, a study by Reza et al. [112] examined two struvites on sandy soil: struvite pre-treated with irradiation and air-dried struvite were tested on sudan grass and compared against fused superphosphate (FSP). The plant yield from both struvites (250 ± 7 g/pot; 225 ± 17 g/pot) was found to be similar to that of FSP (230 ± 10 g/pot). Similarly, Bogdan et al. [69] examined the effects of struvite on perennial ryegrass and found similar (15 ± 2 mg/pot) P uptake compared to TSP (13 ± 1 mg/pot) under sandy soil conditions. However, they noted that the uptake of P from struvite increased in the later stages of the experiment (>50% uptake in last 3 months before harvest), unlike TSP, indicating a pattern of slow release of phosphorus from struvite. Similarly, Robles-Aguilar et al. [113] investigated the effects of struvite on blue lupine grown in sandy soil. They found no significant differences in plant biomass under any pH conditions between struvite (0.18–0.75 g/pot) and synthetic fertilizer (0.19–0.82 g/pot). However, P uptake with struvite was significantly greater (11.8 and 22.4 mg/g DM) than that with synthetic fertilizer (6.6 and 10.2 mg/g DM) in both acidic and alkaline conditions. Interestingly, there was no difference in P uptake efficiency under neutral pH conditions (5.6 mg/g DM to 4.6 mg/g DM).

3.5.2. Struvite Derived from Manure

In a study by Gong et al. [114], an MAP–struvite mixture derived from cattle manure was evaluated across three different vegetative crops (amaranth, barley, and swamp cabbage) and its performance was compared against KCl and control. The results consistently showed that the MAP–struvite mixture outperformed the other treatments across all crops, exhibiting improvements ranging from 40 to 70% in some cases. In a study by Rech et al. [115], struvite derived from swine manure and chicken manure was tested on wheat and soybean as the test crops in a pot experiment. For both crops, the P uptake was

found to be around 20–25% lower and dry yield was also lower by 10–15% compared to with TSP. The lower P uptake and dry yield observed with struvite compared to TSP could be attributed to two main factors. Firstly, TSP has higher solubility than struvite, allowing for more immediate availability of P [115]. Secondly, the larger root system observed with TSP-treated plants may be due to the greater diffusion of P from the source, encouraging root spread. This is supported by the higher root biomass observed with TSP, which was 10–15% higher compared to struvite [115]. In another study by Liu et al. [116], struvite derived from animal urine was assessed on bird rapeseed across two different soil types: cinnamon soil and paddy soil. The experiment involved two conditions: one with normal irrigation water and one with acidic water (pH 6). In the former case, struvite (cinnamon: 0.8 ± 0.1 g/pot; paddy: 0.8 ± 0.05 g/pot) performed similarly to calcium superphosphate (CSP) in both soil types (cinnamon: 0.9 ± 0.3 g/pot; paddy: 0.7 ± 0.1 g/pot). However, when irrigated with acidic water, struvite showed significantly better performance (cinnamon: 1.1 ± 0.4 g/pot; paddy: 1.1 ± 0.3 g/pot) in both soil types compared to CSP. This might be due to the higher P availability from struvite in acidic soils.

Szymańska et al. [117,118] evaluated cattle manure-derived struvite in greenhouse pot trials using two contrasting soils—silty loam and loamy sand. Compared to the negative control, maize and grass yields increased by 66% and 94%, respectively, across both soil textures, highlighting struvite's agronomic effectiveness. Although the performance of struvite for maize ($\text{Yield}_{\text{struvite}}/\text{Yield}_{\text{AP}} = 0.9$) was comparable to that of ammonium phosphate, its performance for grass biomass exceeded that of ammonium phosphate ($\text{Yield}_{\text{struvite}}/\text{Yield}_{\text{AP}} = 1.2$), emphasizing crop-specific benefits. In another study by Robles-Aguilar et al. [33], pansy was tested, and struvite recovered from digested manure (3.4 ± 0.4 g/pot) was found to have a similar yield to TSP (3.5 ± 0.4 g/pot).

Manure and urine-derived struvite generally performed comparably to synthetic P fertilizers across a range of crops, though outcomes varied with soil type and crop physiology. Positive responses were observed when struvite was combined with soluble P sources or applied under acidic conditions, which enhance its solubility [114,116]. While cereals like wheat often showed slightly lower uptake relative to TSP [115], grasses and leafy vegetables tended to benefit more, with biomass sometimes exceeding that of conventional fertilizers [117,118].

Overall, both wastewater and manure-derived struvite have shown comparable performance to conventional P fertilizers, though efficiency often depends on soil type, crop physiology, and water chemistry. Its slow-release characteristics are particularly beneficial in sandy or acidic soils, where nutrient losses from leaching are typically high, and in forage systems where sustained nutrient availability supports biomass growth. From a practical standpoint, struvite recycling offers farmers a renewable, locally available P source that reduces dependence on imported mineral fertilizers such as TSP or MAP. At the same time, its recovery from wastewater and manure treatment systems contributes to nutrient circularity and reduces the risk of eutrophication. Wider field trials under diverse pedoclimatic conditions remain essential, but current evidence positions struvite as a viable component of sustainable P management strategies in line with EU circular economy and fertilizer directives.

3.6. Biostimulants

Biostimulants are substances known to enhance nutrient uptake, stress tolerance, growth, and productivity when applied to plants or soil. Biostimulants are commonly administered to either the roots or leaves of plants, functioning by triggering innate plant functions like photosynthesis, germination, flowering, pollination, and senescence, or by improving the plant's capacity to assimilate and utilize nutrients [119,120]. They can

be either organic or synthetic and are often derived from waste substances (Figure 4). The manufacturing process typically involves the extraction, fermentation, or synthesis of active compounds, followed by formulation for application on crops. Major types of plant biostimulants include seaweed extracts, humic and fulvic acids, biostimulants based on beneficial microorganisms, amino acids, plant extracts, and other organic-based compounds [121].

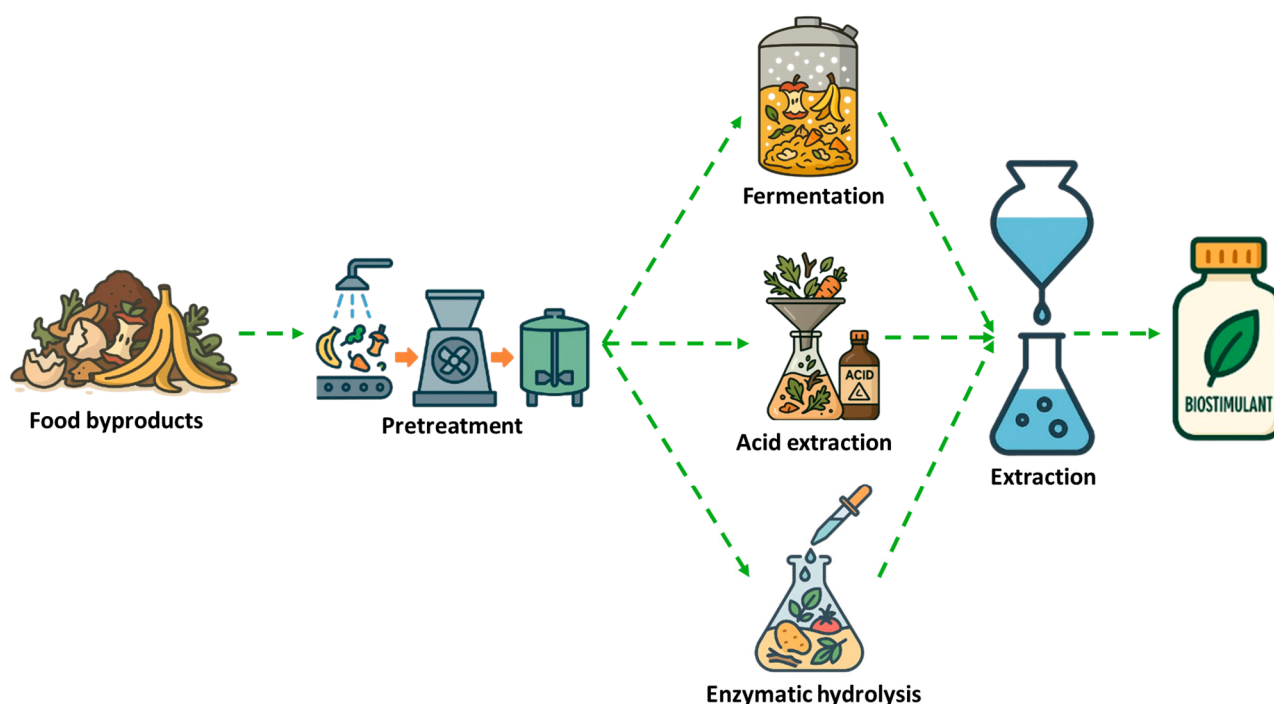


Figure 4. Schematic illustration of biostimulant production from organic waste streams.

The agronomic performance of biostimulants for stress mitigation has been scarcely studied in previous years. In a study by Shrivastava et al. [122], protein hydrolysate-based biostimulants derived from algae cultivated on the retentate of pig slurry were tested on Swiss chard. The study examined the effect of temperature stress on Swiss chard and found that the biostimulant mitigated stress effectively, leading to similar biomass performance (12.8 ± 1.71 g/pot) compared to non-stressed conditions (13.17 ± 1.18 g/pot). A similar study by Sleighter et al. [123] tested the effect of natural organic matter-based biostimulants in 21 field/pot trials across a variety of crops on mitigating salt and drought stresses. The results from the 21 trials highlighted that stress mitigation due to the addition of natural organic matter-based biostimulants improved plant health and consistently increased yields in comparison to stressed conditions ($\text{Yield}_{\text{stress+biostimulants}} / \text{Yield}_{\text{stress}} = 1.3\text{--}2.2$). Another study by Gedeon et al. [124] examined the impact of biostimulants on salt stress in tomato plants through a pot-based experiment. Utilizing humic acid and fulvic acid-based extracts as biostimulants, the study observed a mitigation of stress effects when biostimulants were applied (8.69 ± 0.41 g/pot), resulting in yields similar to those obtained with the negative control under non-stress conditions (8.71 ± 0.58 g/pot). The mitigation of salt stress in plants due to biostimulants could indeed be attributed to the presence of exogenous polysaccharides which could increase plant tolerance to salt stress by enhancing the antioxidant system and regulating intracellular ion concentration [125].

Additionally, biostimulants can be utilized to enhance crop quality and increase yield even in the absence of stress conditions. In a study by Fasani et al. [126], a protein-hydrolysate biostimulant produced from brewery spent grain (BSG) via enzymatic hydrolysis enhanced root biomass by 15% and increased wheat grain nutrient concentrations (Ca,

Mg, K) without reducing overall yield. Furthermore, in a study by Izquierdo et al. [127], a humic biostimulant derived from vermicompost crop residues increased grain yield by 7.4% on average; 93% of trials showed positive responses (mean +8.5%), driven primarily by 6.3–10.5% gains in panicle density and grains per panicle. Also, in a study by Chehade et al. [128], biostimulants derived from the extract of fennel/lemon processing residues and brewer's spent grain were tested on tomatoes in a pot trial. The study revealed a significantly positive effect on fresh fruit yield for all biostimulant treatments, with 14–35% higher total yields compared to negative control treatments. Regarding quality, the plants treated with biostimulants exhibited 10–30% higher levels of vitamin C and phenol content compared to the negative control. In the study by Gurmani et al. [129], biostimulants derived from Moringa leaf and seaweed extracts were tested on oats. The dry matter yield was found to be higher with both extracts (155 ± 9 g/pot; 170 ± 7 g/pot) compared to the negative control conditions (75 ± 5 g/pot). In terms of quality, the biostimulants led to 10–25% higher levels of crude protein and ash content compared to the negative control. However, the crude fiber content was nearly 20% lower in the biostimulant-treated oats compared to the negative control.

In the study by Ngoroyemoto et al. [130], vermicompost leachate and seaweed extract were tested as biostimulants on smooth pigweed. The results showed that the biostimulants led to higher performance in terms of fresh yield (20–30%), chlorophyll (10–40%), protein (45–115%), and carotenoid content (30–35%) compared to the negative control conditions. Similarly, a study by Rouphael et al. [121] tested spinach using three biostimulants derived from legumes, seaweed, and a mixture of vegetal oils, herbal, and seaweed. A 40–60% higher dry yield and 10–15% higher protein content were observed in plots treated with biostimulants. In another study by Mzibra et al. [131] testing the performance of five Moroccan seaweed-based biostimulants on tomato plants, it was found that plants treated with biostimulants exhibited increases in yields of 25% to 90% compared to the negative control. Additionally, in terms of fruit quality parameters, the lycopene content was found to be 33% higher in plants treated with biostimulants. In another study by Rajendran et al. [132], Rockweed algae extract was tested as a biostimulant on sweet pepper plants, resulting in higher plant yield (29.2 g/pot) compared to negative control conditions (14.3 g/pot). Additionally, biostimulants exhibited higher total amino acid content (26 ± 6 mg/g) and increased chlorophyll levels (65 ± 7 SPAD) compared to the control (16 ± 6 mg/g; 58 ± 5 SPAD). The consistent better performance of seaweed extracts in terms of plant yield and parameters may be attributed to several factors. Firstly, the increased efficiency of the extracts could be linked to enhanced availability of active ingredients and improved responsiveness of plants to these components [83]. Seaweed extracts contain various essential elements such as macro and micro nutrients, amino acids, vitamins, cytotoxins, auxins, and abscisic acid-like growth substances, which interact and influence the growth activity of treated plants [133]. These components likely play a role in enhancing plant growth and crop yield by triggering various plant responses without the need for additional growth regulators. The observed effects on plant growth may be correlated with the presence of these components, which enhance the responsiveness of plants to seaweed extract treatments.

In contrast to the consistent performance observed in some crops, a study by Wades and Dziugiel [134] on seaweed extracts from rockweed, sea bamboo, and liquid humic extract with early potatoes showed equal performance in terms of dry yield and chlorophyll content over a three-year period. The authors did not specify the reasons explaining seaweed extract performance, but it is possible that variability in field conditions over the three-year period, along with the presence of exogenous auxin and cytokinins in the seaweed extract, could have influenced the results. Typically, these substances increase leaf

area [135], but they may not have affected the amount of light absorbed by that leaf area throughout the year, which resulted in equal performance to the negative control.

Overall, these findings indicate that biostimulants not only enhance yield and quality but also play a critical role in helping crops tolerate stresses such as heat, drought, and salinity. Their practical relevance lies in providing farmers with low-impact tools to stabilize production under variable climates, reduce reliance on synthetic inputs, and improve the nutritional quality of produce. This positions biostimulants as a valuable component of sustainable and climate-resilient agriculture.

3.7. Sustainability and Market Implications

Waste-derived fertilizers such as ammonium salts, struvite, biochar, and biostimulants offer both environmental and economic co-benefits compared to their synthetic counterparts (Figure 5). For instance, a cradle-to-grave life cycle assessment found that recovered AS from urban wastewater exhibits lower environmental impacts across most categories compared to industrially produced AS, particularly when sourced from urine separation systems [136]. Similarly, full-scale field assessments using digestate-derived AS in maize production reported equal ammonia availability and no accumulation of pollutants, while reducing reliance on synthetic N [26]. Additionally, Rietra et al. [18] observed that AS produced via air-stripping techniques yields significantly lower N₂O emissions than calcium ammonium nitrate plus sulfur in grassland trials under wet conditions, highlighting a potential reduction in greenhouse gas emissions [18]. Economically, the social cost of nitrogen pollution—such as losses through volatilization, eutrophication, or greenhouse gas emissions—is estimated at 32–35 USD per kg of N released, underscoring the high externalities of inefficient fertilization [137]. While direct financial returns on recovered AS remain underexplored, these avoided external costs and reduced energy dependence align with broader sustainability goals.

Additionally, techno-economic analysis of biochar production from sugarcane bagasse in Brazil showed a breakeven within ~7.5 years and an internal rate of return of 18%, with profitability strongly dependent on carbon credit values above 120 USD t^{−1} CO₂e [138]. Life cycle assessments further indicate that biochar's sequestration costs (200–584 USD t^{−1} CO₂) are competitive with engineered removal technologies while simultaneously improving soil water retention and reducing nutrient leaching [139,140]. Struvite recovery systems also deliver environmental gains by lowering eutrophication potential and substituting energy-intensive phosphate fertilizers, with electrochemically precipitated struvite achieving higher P uptake and yields than conventional fertilizers [108,115]. On the economic side, farm-scale struvite recovery has been shown to produce net savings of 65–100% relative to imported mineral P fertilizers [67].

On the policy front, recovering ammonium salts supports circular economy objectives and aligns with current EU fertilizing product regulations that permit the use of recovered N sources (Appendix A). However, widespread adoption faces obstacles such as complex regulatory approval, upfront infrastructure costs, and concerns over application-specific volatilization risks, particularly when using liquid AS [30]. Tailoring application techniques (e.g., trailing shoe applicators) and addressing environmental losses will be critical to translating recovered ammonium salts' environmental promise into farmer uptake. Furthermore, adoption is increasingly shaped by policy and regulatory frameworks of the EU FPR [7], which now recognizes recovered nutrients (Figure 6), while nutrient runoff restrictions under the Nitrates Directive and circular economy strategies provide additional market pull [11]. Nonetheless, barriers remain, including high upfront investment costs, certification complexity, and uncertainties regarding product stability and farmer acceptance [15]. Collectively, these insights underline that while recovered fertilizers already demonstrate

clear sustainability advantages, supportive carbon markets and regulatory alignment will be decisive for scaling their adoption.

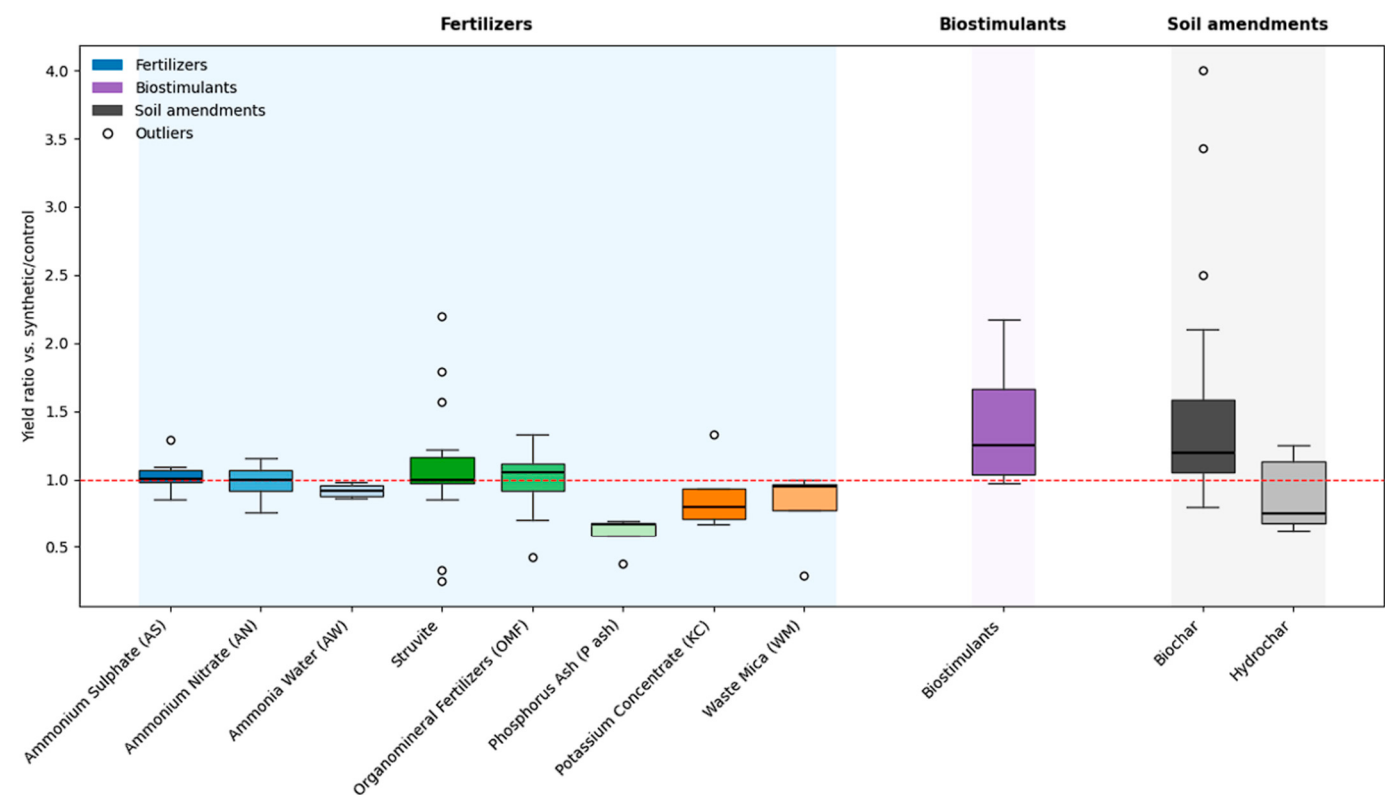


Figure 5. Yield ratios of waste-derived fertilizers, biostimulants, and soil amendments compared to synthetic or control treatments. Boxplots show median, interquartile range, and outliers; red dashed line indicates parity with synthetic/control.

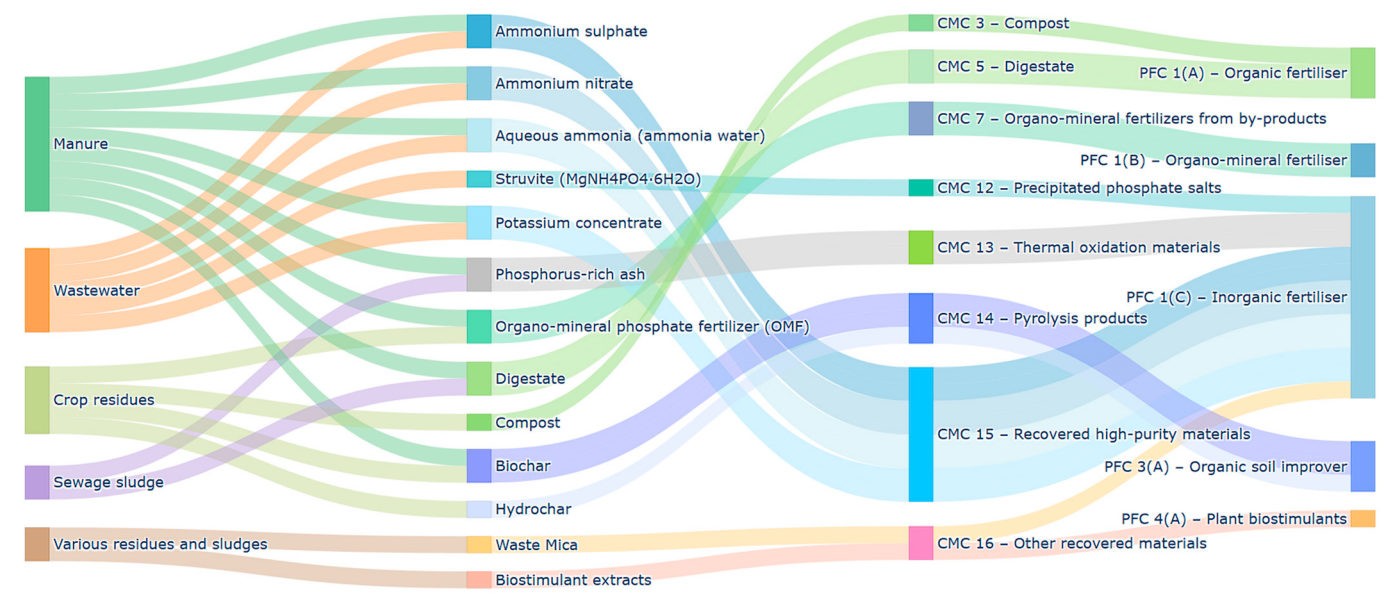


Figure 6. A Sankey diagram showing the transformation of manure, wastewater, crop residues, sewage sludge, and other residues into waste-derived fertilizers and their corresponding EU Fertilising Products Regulation classifications (CMCs and PFCs).

4. Conclusions

The past decade has observed significant progress in the utilization of waste for fertilizer production, marked by the development and implementation of various nutrient recovery technologies. These technologies have shown efficacy in extracting nutrients from diverse sources such as crop residues, manure, and wastewater streams. However, while some waste-recovered fertilizers have undergone thorough agronomic assessment, notable gaps persist in research, particularly concerning ammonium-based salts and recovered potassium products. Moreover, field validation studies have been limited compared to greenhouse or laboratory experiments, indicating a need for more extensive trials to evaluate real-world effectiveness.

In studies focusing on N sources, ammonium salts derived through the stripping–stripping method have shown comparable efficiency to synthetic fertilizers like CAN and urea, with consistent trends in yields and nitrogen uptake observed over the years. Similarly, for P, struvite recovery from wastewater and manure has demonstrated good agronomic performance, alongside biochar. However, the debate persists over whether biochar should primarily serve as a phosphorus source or soil improver. Regarding K, K concentrate has shown performance similar to that of synthetic KCl. However, a major issue across these studies is the state of the recovered fertilizer, often being liquid-based, posing challenges for transportation and application due to specialized machinery requirements.

Despite these challenges, waste-recovered nutrients have demonstrated promising agronomic efficiency and exhibit high potential for substituting synthetic fertilizers in agriculture. Nonetheless, a comprehensive evaluation of the environmental implications of bio-based fertilizers is still lacking. To address these gaps, further research is recommended, focusing on agronomic performance, field validation, and environmental impact assessments of waste-recovered fertilizers. Collaborative efforts between researchers, agricultural practitioners, and policymakers are crucial to facilitate the development and adoption of sustainable waste-to-fertilizer technologies. Long-term studies are warranted to assess environmental implications such as nutrient leaching, greenhouse gas emissions, and soil health associated with the use of bio-based fertilizers. Additionally, education and outreach initiatives should be undertaken to raise awareness among farmers and stakeholders about the benefits and challenges of transitioning to waste-derived fertilizers. Policy support and incentives may also be necessary to promote sustainable fertilizer practices and encourage investment in waste-to-fertilizer technologies.

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Abbreviations

The following abbreviations are used in this manuscript:

AN	Ammonium Nitrate
AS	Ammonium Sulfate
AW	Ammonia Water (Ammonium hydroxide, NH_4OH)
BBF	Bio-Based Fertilizer
CAN	Calcium Ammonium Nitrate
CMC	Component Material Category (EU Fertilizer Regulation)
EC	European Commission
EU	European Union
FAO	Food and Agriculture Organization
FPR	Fertilizer Products Regulation (EU 2019/1009)
IWA	International Water Association
JRC	Joint Research Centre (European Commission)
NDVI	Normalized Difference Vegetation Index
NNI	Nitrogen Nutrition Index
NPK	Nitrogen, Phosphorus, Potassium
OMF	Organo-Mineral Fertilizer
PFC	Product Function Category (EU Fertilizer Regulation)
RENURE	REcovered Nitrogen from manURE
UAV	Unmanned Aerial Vehicle
UNESCO	United Nations Educational, Scientific and Cultural Organization
WWAP	World Water Assessment Programme (UNESCO)

Appendix A. Detailed Classification of EU FPR

Table A1. Classification of the recovered nutrient products discussed in this review under the EU Fertilizer Products Regulation (FPR 2019/1009), showing Component Material Categories (CMCs) and Product Function Categories (PFCs) [7].

Product	Source Stream(s)	CMC (Component Material Category)	PFC (Product Function Category)
Ammonium sulfate	Manure, wastewater	CMC 15 (Recovered high-purity materials)	PFC 1(C)(I)(a)(i)—Solid inorganic macronutrient fertilizer
Ammonium nitrate	Manure, wastewater	CMC 15 (Recovered high-purity materials)	PFC 1(C)(I)(a)(i)—Solid inorganic macronutrient fertilizer
Aqueous ammonia (“ammonia water”)	Manure, wastewater	CMC 15 (Recovered high-purity materials)	PFC 1(C)(I)(a)(ii)—Liquid inorganic macronutrient fertilizer
Struvite ($\text{MgNH}_4\text{PO}_4 \cdot 6\text{H}_2\text{O}$)	Municipal wastewater	CMC 12 (Precipitated phosphate salts)	PFC 1(C)(I)(a)(i)—Solid inorganic macronutrient fertilizer
Organo-mineral phosphate fertilizer (OMF)	Crop residues, manure	CMC 7 (Organo-mineral fertilizers from by-products)	PFC 1(C)(I)(b)—Solid mineral/organo-mineral macronutrient fertilizer
Phosphorus-rich ash	Sewage sludge ash, manure ash	CMC 13 (Thermal oxidation materials)	PFC 1(C)(I)(b)—Solid synthetic fertilizer
Potassium concentrate	Manure, wastewater	CMC 15 (Recovered high-purity materials)	PFC 1(C)(I)(a)(ii)—Liquid inorganic macronutrient fertilizer
Waste Mica	Mica mining waste	CMC 16 (Other recovered minerals)	PFC 1(C)(I)(b)—Solid synthetic fertilizer

Table A1. Cont.

Product	Source Stream(s)	CMC (Component Material Category)	PFC (Product Function Category)
Biochar	Crop residues, manure	CMC 14 (Pyrolysis products)	PFC 3(A)—Organic soil improver
Hydrochar	Crop residues	CMC 14 (Pyrolysis products)	PFC 3(A)—Organic soil improver
Compost	Crop residues, food waste	CMC 3 (Compost)	PFC 1(A)—Organic fertilizer
Digestate	Manure, sewage sludge	CMC 5 (Digestate)	PFC 1(A)—Organic fertilizer
Biostimulant extracts	Various residues and sludges	CMC 16 (Other recovered organics)	PFC 4(A)—Plant biostimulant

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