Chinese Chemical Letters 37 (2026) 111431

Contents lists available at ScienceDirect

Chinese Chemical Letters

journal homepage: www.elsevier.com/locate/cclet

Recovery of nitrogen and phosphorus in sewage sludge treatment technologies

Lu Liu[a], Yihe Wang[a], Jialin Wang[a], Chao He[b,*], Jun Zhang[a,*]

[a] State Key Lab of Urban Water Resource and Environment, National Engineering Research Center for Safe Disposal and Resource Recovery of Sludges, School of Environment, Harbin Institute of Technology, Harbin 150090, China
[b] Faculty of Engineering and Natural Sciences, Tampere University, Tampere 33720, Finland

ARTICLE INFO

Article history:
Received 4 December 2024
Revised 11 April 2025
Accepted 8 June 2025
Available online 16 June 2025

Keywords:
Sewage sludge
Nitrogen
Phosphorus
Treatment technology
Resource recovery

ABSTRACT

Sewage sludge (SS) is a by-product of wastewater treatment. Recovering resources from SS, particularly nitrogen (N) and phosphorus (P), is emerging as a crucial approach to promoting carbon neutrality within the treatment sector. This need necessitates developing methods that not only recover these vital nutrients but also mitigate the release of nitrogenous pollutants. Our review addresses the recovery of N and P from SS, aiming to reduce the dissemination of harmful nitrogen compounds into the environment. We provide a comprehensive analysis of techniques ranging from ultrasonic treatment and aerobic/anaerobic digestion to thermochemical conversion methods such as incineration, pyrolysis, and gasification. We also evaluate strategies like bio-enhanced phosphorus recovery and electrochemical methods, with the dual goal of diminishing nitrogen emissions and reclaiming phosphorus from SS. The review synthesizes advanced techniques and strategies to support emission control and resource recovery, offering a comprehensive guide for advancing SS treatment technologies.

© 2025 Published by Elsevier B.V. on behalf of Chinese Chemical Society and Institute of Materia Medica, Chinese Academy of Medical Sciences.

1. Introduction

The acceleration of urbanization and increasingly stringent water discharge regulations have significantly increased sewage sludge (SS) production, drawing considerable attention in recent years [1]. The annual production of municipal SS exhibits significant regional disparities: The United States generates approximately 6 million tons of dry matter [2], China produces about 30 million tons [3], Japan outputs 2.34 million tons of dry matter [4], and Australia contributes 0.37 million tons [5]. During SS treatment, hazardous elements are released, leading to environmental pollution. This includes the emission of highly reactive nitrogen oxides (NO_x) during heat treatment, which contribute to photochemical smog, acid rain, ozone layer depletion, and an intensified greenhouse effect, causing significant secondary pollution [6–8]. The tar produced by SS pyrolysis can degrade the quality of pyrolysis gas and corrode equipment and piping systems [9–11]. Direct landfilling of SS generates leachate with high concentrations of nitrogen (N) and phosphorus (P), which seeps into groundwater and subsequently enters surface water bodies *via* subterranean and surface runoff, contributing to harmful environmental impacts such as eutrophication [12,13]. The advantages and disadvantages of these sludge treatment methods are summarized in Table S1 (Supporting information).

In addition to proper SS treatment, consideration should be given to the environmental benefits of pollution reduction and the economic benefits of resource utilization. N and P are essential nutrients for living organisms, sustaining all life on Earth [14,15]. P regulates algal growth and plankton species distribution by limiting biological subsets in the ecosystem [16,17]. Furthermore, N provides raw materials for industrial production, fertilizers for agriculture and aquaculture, and food and energy for households [18,19]. P is used in industrial ores, animal feed, and as a fertilizer for crops [17]. Rapid urbanization and excessive human disturbance have greatly promoted the flow of N and P from the socio-economic system to the ecological environment, accelerating the N and P cycles [20]. Fig. 1 illustrates the N and P cycles in human production activities. Therefore, reducing the flow of N and P from SS into the ecological environment has become a focus for sustainable resource management and a circular economy.

Efforts have been made to explore strategies to enhance resource reclamation and pollutant control during sludge treatment and disposal in recent years (Figs. S1-S3 in Supporting information). However, limited reviews on SS resource recovery highlight

* Corresponding authors.
E-mail addresses: hitsunyboy@126.com (J. Zhang), chao.he@tuni.fi (C. He).

https://doi.org/10.1016/j.cclet.2025.111431
1001-8417/© 2025 Published by Elsevier B.V. on behalf of Chinese Chemical Society and Institute of Materia Medica, Chinese Academy of Medical Sciences.

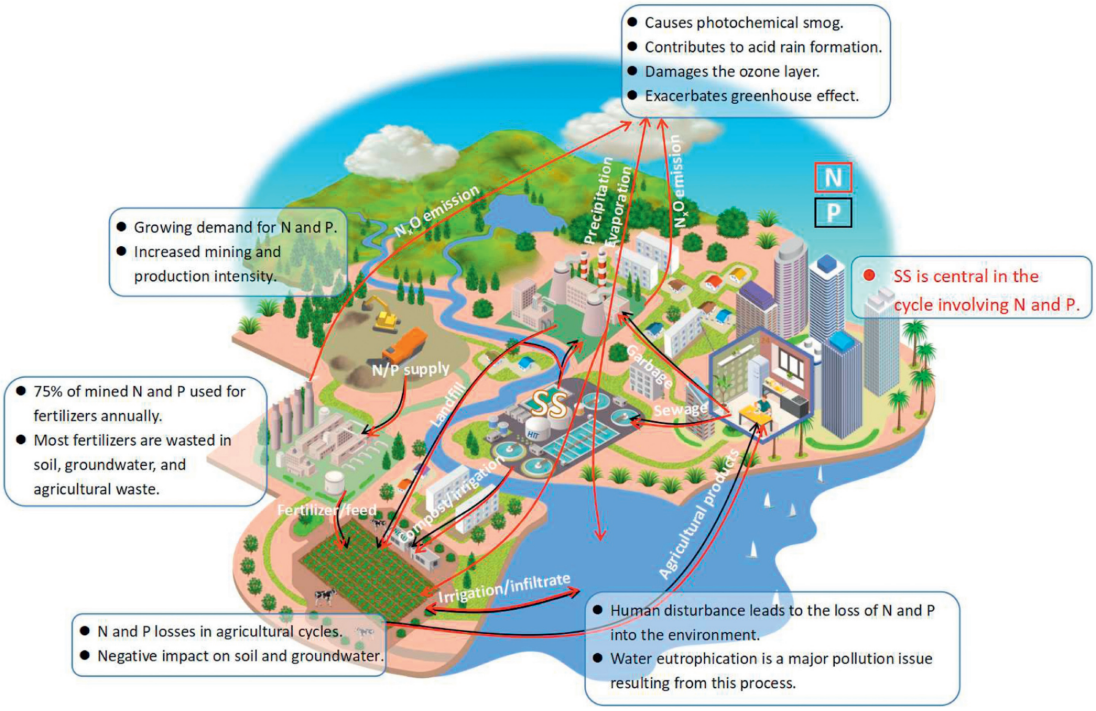


Fig. 1. The N and P cycles in human production activities.

the complex behaviors of nitrogen and phosphorus elements—such as their release, distribution, and transformation—and the associated environmental pollution, hindering the development of effective recovery techniques [21]. The significance of resource recovery and sustainable development is increasingly recognized and practiced. Thus, up-to-date information and comprehensive summaries are crucial to highlight the technical and economic feasibility of different resource recovery technologies for SS. This review covers three main parts: (1) the release of nitrogen and phosphorus in SS; (2) nitrogen conversion in SS treatment and pollutant emission control; and (3) phosphorus recovery from SS. This review aims to provide valuable information to advance the N and P recovery industry and inspire future research toward pollution abatement.

2. Forms and characteristics of N and P present in SS

2.1. Nitrogen

The N content in SS ranges from 2.80 % to 8.06 % of total solids (TS) [22,23]. N in SS exists in both organic and inorganic forms. Organic N includes pyrrole-N, pyridine-N, protein-N, and amine-N [24], while inorganic N consists of ammonia N (NH_3-N), nitrite N (NO_2-N), and nitrate N (NO_3-N) [25]. Organic N accounts for 99.35 % of the total N in SS. The primary organic N components are labile protein-N and pyrrole-N, comprising 1.68 % TS and 2.53 % TS of SS, respectively [26]. The main amino acids in SS proteins constitute 88.07 % of the total amino acids, arranged in the following order: glutamic acid > leucine > arginine > lysine > glycine > valine > tyrosine > serine > isoleucine > aspartic acid [27]. This sequence is primarily influenced by the source of SS. Therefore, different human activities across regions may lead to variations in the predominant amino acid species in SS. Various transformation modes and reaction products of N occur in different amino acids during fermentation and pyrolysis processes [28,29]. In SS, the interaction between heavy metals, organic pollutants, and nitrogen compounds through organometallic complexation significantly affects their bioavailability and mobility in the environment [30,31]. These interactions can alter the forms of nitrogen, making them either more accessible for recovery or more challenging to extract and utilize. In summary, the complex and varied nature of nitrogen in SS necessitates a multifaceted approach to recovery.

2.2. Phosphorus

The P content in raw SS varies significantly, ranging from 0.1 % to 14 % on a TS basis, depending on the characteristics of the original SS [32,33]. Accurate assessment of P content is crucial for effective recovery strategies, as treatment methodologies can further influence P availability for recovery. In urban SS, P concentrations range from 2.21 % to 17.7 % as P_2O_5 [34], whereas post-incineration industrial sludge ash contains 15 % P_2O_5, similar to the lowest phosphorus levels found in phosphate ores [35]. P tends to deposit in an inorganic phase or concentrate into an organic form. Among the inorganic P species, PO_4^{3-} compounds predominate [36], while organophosphorus consists of various organic compounds, such as deoxyribonucleic acid, ribonucleic acid, adenosine triphosphate, adenosine diphosphate, adenosine monophosphate, monoester-P, diester-P, phospholipids, and phytic acid [37,38]. Organic P in SS primarily originates from microbial metabolism and municipal waste, including food and detergent residues. P in SS mainly exists in microbial cells and extracellular polymeric substances (EPS) in the form of polyphosphates, orthophosphates, and organic P [39]. Inorganic P in SS is typically formed by the combination of phosphate and metal ions, mainly from chemical coagulants and raw wastewater. In this context, inorganic P can be divided into chemically precipitated inorganic P (Fe-P and Al-P, 45 %–78 % of total inorganic P) and inorganic P (Ca-P and Mg-P, 14 %–37 % of total inorganic P) precipitated from raw wastewater [40,41]. The diverse forms of phosphorus in SS, both organic and inorganic, underscore the complexity of its recovery and reuse. The presence of various organic P compounds, such as nucleic acids and phospholipids, predominantly originating from microbial

metabolism and municipal waste, presents challenges in terms of separation and extraction.

3. Fate of N and P in treatment and disposal of SS

3.1. Physicochemical techniques

3.1.1. Freeze-thaw

Sludge freezing-thawing technology is an efficient pretreatment method involving the freezing and thawing of sludge at low temperatures. After the freeze-thaw process, SS is dehydrated and densified, reducing the cost of *ex-situ* processing [42]. During freezing, water in the sludge forms ice crystals, damaging the flocculation structure and cell membranes, leading to the release of cell contents during thawing, and rapidly killing pathogens in the freezing phase. The average dehydration rate of slow freezing (−10 °C and −20 °C) is higher than that of rapid freezing (−80 °C) [43]. Freeze-thaw treatment increases the NH_4^+-N concentration by 2–8 times and the PO_4^{3-}-P concentration by 1.5–2.5 times [44]. The drying rate of SS improves by 14.38 %–30.97 % compared to untreated sludge [44]. This improvement is likely due to cell rupture and the release of intracellular water into the SS supernatant, converting intracellular water to free water. Post freeze-thaw treatment, the pollutant removal rate during anaerobic fermentation of SS significantly improves, with average biogas production reaching 1.31 m^3/kg, approximately 1.5 times the original sludge value [45]. Clearly, freeze-thaw technology not only enhances the release of N and P from SS but also boosts the efficiency of subsequent SS treatment and N and P recovery. However, freeze-thaw technology has higher requirements for low-temperature conditions, making it more suitable for cold regions.

3.1.2. Ultrasonication

The use of ultrasonics to destroy the flocculent structure of SS results in the breakdown of flocs and lysis of cells, leading to the dissolution, hydrolysis, and release of N and P components [46]. Ultrasonic treatment offers high efficiency, simple equipment, and convenient operation. Optimal ultrasonic intensity is crucial in the N and P release process. At an ultrasonic energy density of 720 W/L, the N and P concentrations in the supernatant of treated samples were approximately 2.1 times those of the control group. Beyond 720 W/L, no significant changes in N and P concentrations were observed [47]. However, ultrasonic treatment is energy-intensive and demands high equipment and energy consumption, limiting its scalability and practical implementation [48]. To address these limitations, coupling ultrasonic treatment with acids, alkalis, surfactants, and oxidants has emerged. Sodium citrate-ultrasonic and NaOH-ultrasonic combined pretreatments have significantly disrupted the sludge floc structure [49]. Surfactant-ultrasonic treatment further reduces the low surface tension of SS, enhances the destruction of non-covalent bonds between EPS and cells, and increases the release of intracellular polymers [50]. Short-term treatment with potassium permanganate-ultrasound increased supernatant total nitrogen (TN) and soluble protein by 1029.4 % and 2996.6 %, respectively [51]. Under 0.8 % H_2O_2 and 70 W ultrasonic treatment, sludge mass and volume reduction rates exceeded 50 %, promoting the release of proteins in the supernatant [52]. Using ultrasonically treated anaerobic sludge supernatant for irrigating wheat fields significantly reduces the need for synthetic fertilizers, cutting urea usage by 122.25 kg/ha and superphosphate by 6.75 kg/ha [53]. This reduction results in a significant decrease in CO_2 emissions, amounting to 699.40 kg CO_2eq/ha after life cycle assessment of fertilizer production and application. Given China's vast cropland area, adopting this practice could potentially reduce greenhouse gas emissions by over 82 million tonnes annually, offering substantial agricultural and environmental benefits [54]. All the aforementioned coupling processes enhance the breakdown of SS and promote the release of N and P into the supernatant. To further improve ultrasonic efficiency and reduce energy input, future coupling processes should extend beyond acids, bases, surfactants, and oxidants, incorporating methods such as aeration-ultrasonic, ozone-ultrasonic, and thermal-ultrasonic treatments.

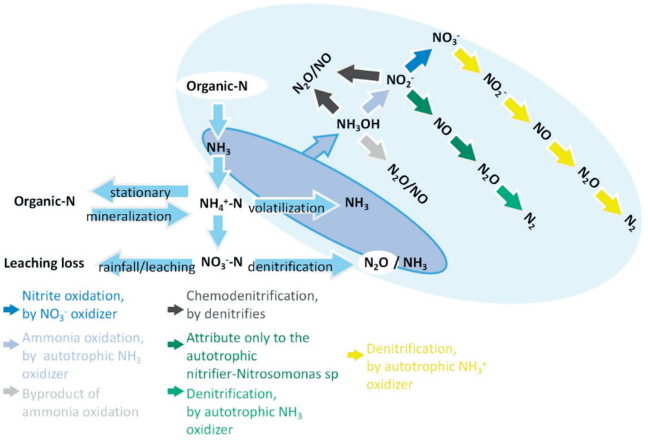


Fig. 2. Transformation pathways of N and potential pathways for N_xO production in aerobic biological treatment of SS.

3.1.3. Ozonation

Ozone oxidative degradation disrupts the flocculent structure of sludge, exposing a significant number of microbial cells [55]. Injecting 380 mg/L of ozone for 2 min increased the TN concentration in the SS supernatant from 8 mg/L to 21 mg/L and the total phosphorus concentration from 0.3 mg/L to 2.1 mg/L [56]. Compared to methods like ultrasonic treatment and thermal hydrolysis, the cost of ozonation for sludge disintegration is relatively modest, making it a cost-effective option for enhancing nutrient recovery. Under the same ozone conditions, the total N concentration released into the supernatant by the micro-bubble system was more than twice that of the bubble system [57]. The effects of ozone on SS vary with pH: in acidic environments, ozonation enhances nutrient release, while in alkaline environments, it enhances SS dissolution [58,59]. Additionally, adding iron, calcium, and magnesium salts can promote the release of phosphorus into the supernatant, forming high concentrations of phosphate [57,58]. Despite significant advances in ozone oxidation technology, challenges remain in its full implementation. While the cost of ozonation for sludge disintegration is relatively low, the cost of generating devices and the increased daily maintenance expenses are significant constraints.

3.2. Biological processes

3.2.1. Aerobic composting

The transformation pathways of N and potential pathways for nitrous oxide (N_xO) production in SS aerobic biological treatment are shown in Fig. 2. Aerobic composting achieves higher inorganic N removal from SS, greater sludge degradation to release ammonia, and improved SS N removal capacity [60]. Organic phosphorus can be oxidized into phosphate [61]. The high-temperature phase is critical for efficient aerobic composting of sludge, and composting SS alone may limit temperature increase, reducing microbial activity and inhibiting N and P release [62]. Co-composting SS with kitchen waste, while maintaining a C/N ratio of 25, effectively degrades proteins [63]. The microbial diversity and richness in mixed SS and kitchen waste compost are higher than in SS and

kitchen waste composted separately, maximizing the abundance and metabolic activity of functional microorganisms and facilitating N and P release [63]. Zero-valent iron has been used effectively to enhance the anaerobic digestion of SS, and recent studies show it is also suitable for aerobic composting of SS [60]. In aerobic composting, zero-valent iron promotes the hydrolysis and decomposition of SS in a dose-dependent manner. However, estrogens such as 17β-estradiol and 17α-ethinylestradiol have inhibitory effects during aerobic composting of SS [64]. When 20 μg/L of 17β-estradiol and 17α-ethinylestradiol were added to the reactor, ammonium removal dropped dramatically from 97 % to 58 %. Therefore, co-composting SS with livestock manure should be avoided. Future research on aerobic composting of SS should focus on improving composting efficiency and exploring chemicals that inhibit the process.

3.2.2. Anaerobic digestion

P degradation and release during anaerobic digestion depend on the hydrolysis of organic P into soluble P. However, during anaerobic digestion, the organic structure of P is easily destroyed, leading to re-precipitation or adsorption on the solid surface with other cations, and only 10 % of the phosphorus is emitted with the biogas [65]. As the sludge retention time (SRT) increases, the content of organic matter and organic nitrogen in the supernatant after acidification also increases [66]. This indicates that the refractory protein in SS increases with longer SRT. When the SRT is $<$10 days, SS is suitable as an internal carbon source and P resource for anaerobic acidification. P released from SS can inhibit the activity of methanogenic bacteria, thereby suppressing methane production [67,68]. Zero-valent iron [69] and Core-shell ZVI@carbon [70] have been shown to alleviate this inhibition, but cannot completely prevent it. Given these constraints, anaerobic digestion becomes less favorable when methane release and recovery are primary objectives. This highlights the significant impact of P transformations during anaerobic digestion on the process and its outcomes. Consequently, future research should investigate how changes in N morphology affect anaerobic digestion, suggesting potential avenues for optimizing process efficiencies and product yields.

3.3. Thermochemical conversion

3.3.1. Conventional thermochemical treatment

Thermochemical treatment technologies, including dry heat chemical treatment (incineration, pyrolysis, and gasification) and wet heat chemical treatment, have become increasingly popular for solid waste treatment [71–73]. During SS incineration, organic P volatilizes, while inorganic P forms stable compounds like $AlPO_4$ and $Ca_2P_2O_7$, with approximately 66.81 %–69.30 % of P concentrated in sludge fly ash [74]. The release of NO_x is a key environmental issue during SS combustion [73]. In SS pyrolysis, N-containing compounds, mainly proteins, decompose into gaseous (30 %–60 %), liquid (15 %–30 %), and solid (25 %–35 %) compounds, producing NH_3 and HCN [75,76]. Higher temperatures promote the release of NH_3 from solid and liquid forms [77]. P remains largely in the solid phase, with organic-P converted into inorganic-P forms like *ortho*-P and *pyro*-P [78,79]. During SS gasification, NO_x precursors are HCN, NH_3, and tar-N, with $>$90 % of P retained as calcium phosphate in the ash [80–82]. Similar findings are observed in the gasification of P-rich biomass [83]. Although dry heat treatment offers advantages like volume reduction, bacterial destruction, and energy recovery, challenges remain, including high energy consumption, long drying times, and product quality. The use of catalysts and additives is gaining attention to improve the process [82].

3.3.2. Hydrothermal processing

The basic principle of hydrothermal treatment is to use high temperatures to destroy sludge flocs and cell structures [84]. This process releases water and other inclusions, improving sludge dehydration [85]. Hydrothermal treatment includes hydrothermal carbonization, liquefaction, oxidation, and gasification, with hydrogen and methane recovery being primary focuses in gasification [86]. Hydrothermal gasification has higher input and energy consumption compared to other treatments. Chemically enhanced thermal hydrolysis, combining caustic soda and low-grade heat, advances disintegration of cell walls, reducing sludge volume, disposal costs, and enhancing biogas production [87]. Hydrothermal carbonization improves fuel quality for thermochemical utilization (Figs. S1 and S2b). At 250 °C, 27.1 % of N remains in hydrochar, while 69.2 % and 6.7 % are found in aqueous and oil products, respectively [88]. Protein-N in hydrochar is converted into heterocyclic-N as temperature increases, with P precipitating as inorganic minerals over time [89]. Unlike pyrolysis, where P exists as polyphosphate, hydrothermal carbonization shows different P behaviors [90]. In hydrothermal liquefaction, by-products like biochar and aqueous products are formed along with biocrude [91]. Increasing temperature converts amine-N to heterocyclic-N, raising N in biocrude. Higher pressures and lower temperatures during liquefaction increase N in the aqueous phase and P in biochar [92,93]. During hydrothermal oxidation, ammonia-N in the aqueous phase increases with the H_2O_2 mass fraction, reaching a maximum of 2100 mg/L at 230 °C [94]. Higher temperatures cause slight decreases in ammonia-N concentration due to weaker thermal stability [95]. Understanding the development, parameters, composition, and product yields is crucial for improving hydrothermal chemical reactions and guiding practical applications. Further research is needed to enhance the feasibility of moist heat chemical treatment in SS processing.

As shown in Table S3 (Supporting information), Physicochemical treatment methods, such as freeze-thaw cycles, ultrasonic treatment, and ozonation, can significantly enhance the release of N and P from SS. However, some methods, such as ultrasound, tend to have high energy consumption. Biological treatments, including aerobic composting and anaerobic digestion, also affect N and P removal, though their efficiency is influenced by various factors, such as co-composting and SS retention time. Thermochemical methods, including incineration, pyrolysis, and hydrothermal treatment, release and transform N and P in SS through different mechanisms. In particular, hydrothermal treatment has shown promise in improving nutrient release.

4. Mitigation strategies for N emissions from treatment and disposal of SS

4.1. Nitrogen emission reduction strategies during SS biochemical treatment

4.1.1. Chemical additive

During composting, N gases like NH_3, N_2, NO_x, and N_2O are produced, with N loss reaching up to 68 %, mainly through ammonia volatilization [96]. Controlling ammonia volatilization and reducing nitrogen loss is crucial. Chemical additives have been shown to be effective in mitigating N loss [97].

Adding carbon sources such as glucose, sucrose, starch, and molasses can improve ammonia assimilation and accelerate the conversion of NH_4^+-N to biological-N, reducing NH_3 and N_2O emissions [98]. For instance, introducing sucrose reduced nitrogen loss by 25.23 %, and adding beet pulp further reduced ammonia emission by 56.7 % [99]. $MgCl_2$ and $FeSO_4$ reduced NH_3 emissions by 58.3 % and 82.9 %, respectively [100]. Acidic additives like lactic acid and citric acid reduced N loss by 4.25 %–14.65 % and did

not negatively affect hygiene [101,102]. Biochar and manganese ore co-addition reduced N_2O emissions by 22.9%–33.7% [103], while zeolite and biochar together reduced NH_3 and N_2O emissions by 58.03%–65.17% and 95.14%–97.28%, respectively [104]. Zeolite and lime compounded reduced N loss by 50% [105]. $MgSO_4$ and KH_2PO_4 enhanced ammonium recovery, reducing nitrogen loss by 62.5% and NH_3 loss by 72.7% [106].

In anaerobic nitrification, N-rich supernatant is produced, which can be treated by nitrification-denitrification. The CO_2 degassing technique with MgO and struvite precipitation recovered approximately 50% of NH_4-N from the supernatant [107]. Conventional struvite precipitation using $MgCl_2$ recovered 88.3% N, while Na-zeolite ion exchange achieved 84.6%, improving to 90.5%–92% with two-step precipitation recovery [108]. Although chemical additives improve SS composting, they increase disposal costs, making cost-effective fillers more beneficial for SS composting efficiency.

4.1.2. Co-composting

Co-composting enhances soil fertility and reduces pollution more effectively than single-component SS composting, aligning with circular development principles [109]. It is cost-effective, merging resource recycling with SS treatment, and reduces reliance on synthetic fertilizers, conserving non-renewable energy [110,111]. This method is particularly beneficial for rural waste management, creating a micro-circulation system when SS is co-composted with meal waste, overcoming limitations of separate composting methods [112]. Co-composting neutralizes pollutants in SS and food waste, improves organic matter in SS, and addresses low pH and high water content in food waste.

Using bagasse as filler, stacking SS and meal waste in a 2:1 ratio promotes glutamine production, while a 4:1 ratio increases ammonia nitrogen concentration in humus to 5.84 mg/g [113]. Modified pine as filler reduces nitrogen loss by 9.40% during co-composting with SS and sawdust, promoting organic matter degradation [114]. Mixing dehydrated SS with spent mushroom substrate (10:3 ratio) and molasses results in a 33.1%–37.3% reduction in NH_3 volatilization, a 17.8%–25.4% reduction in N_2O volatilization, and a 27.2%–32.2% reduction in nitrogen loss [115]. The biomass loosens the structure of SS, increasing porosity, improving oxygen transfer, and reducing NH_3 and N_2O emissions [116,117].

Co-composting also enhances anaerobic digestion, shortening digestion times for SS with disposable diapers, expired food, and sawdust [118,119]. NH_4^+-N concentration in digestate from microalgae and SS co-digestion reached 395 mg/L, and ammonium sulfate was recovered using a hydrophobic polypropylene hollow fiber membrane contactor with 99% recovery efficiency [120]. Co-composting improves microbial activity, reduces nitrogen loss, and enhances metal immobilization in SS compost, improving its quality and safety.

4.1.3. Other biochemical-methods

An advancement in SS aerobic composting is the direct cultivation of grass within it [121]. Pennisetum and Nigella were cultivated and composted in SS with only 5% N loss after decay [122]. NH_4^+ concentration decreased by 98%, while NO_x concentration increased by 9.8 times due to nitrification [123]. Mineral nitrogen from ammonia volatilization, leaching, and absorption was returned in organic form with grass addition for composting. Inoculating thermotolerant nitrifying bacteria during the thermophilic phase reduced NH_3 emissions by 29.7% by converting NH_4^+-N to NO_3^--N [124]. Increased nitrifying bacteria had no negative effect on SS decay efficiency. Hyperthermophilic pretreatment at 120 °C for 30 min increased humification by 123.3% and N retention by 14.2%, promoting humic acid formation and reducing N loss [125,126]. It disrupted sludge EPS, increased protein solubilization by 37.5%, and accelerated decay [125].

Several mitigation strategies including the application of chemical or mineral amendments, co-composting techniques, grass cover establishment, inoculation with thermotolerant nitrifying bacteria, and hyperthermophilic pretreatment have been demonstrated to reduce nitrogen losses, particularly NH_3 emissions, during aerobic sewage sludge composting. In anaerobic SS composting, nitrogen hardly produces gaseous products, with fewer studies on N emission reduction. A simultaneous strategy using post-alkali stripping treatment with potassium ferrate enhanced SS hydrolysis and increased N recovery potential to 8.71 mg NH_4^+-N/g VSS [127]. This offers new solutions for N emission reduction and resource recovery in anaerobic processes.

4.2. Strategies for N emission reduction during thermochemical treatment

Biomass such as SS contains abundant N, which is released as gas-phase products upon heating, particularly during gasification. NH_3 and HCN are the main precursors of NO_x generated during gas combustion. Numerous factors affect N conversion during the thermochemical treatment of SS, including the types of SS, additives, heating rate, temperature, pressure, reaction device, and atmosphere. These factors influence the physical and chemical structure of pyrolysis products and determine the composition and proportion of various nitrogenous products, thereby affecting the degree of nitrogen conversion in sludge. Current research on N emission reduction during the thermochemical treatment of SS is relatively limited and lacks a systematic understanding.

4.2.1. Incineration

In the process of SS incineration, the transport and transformation mechanism of fuel N mainly includes two pathways: (a) transport and transformation of nitrogenous compounds in volatile components, and (b) migration and transformation of N in coke. The formation mechanism of nitrogen oxides (NO and N_2O) involves thousands of reactions. During the incineration of solid fuel, fuel particles rapidly pyrolyze under high temperatures, releasing volatile components, and volatile N forms intermediate products such as NH_3 and HCN, which are precursors of NO_x. Methods for inhibiting NO_x during sludge combustion are primarily divided into two categories: regulating sludge composition by adding adjuvants or mixing with other biomass, and improving incineration equipment. Controlling temperature, air flux, and multi-stage combustion during the incineration process reduces the concentration of NO_x in the soot. The mechanism of nitrogen transformation in the co-combustion process of biomass remains unclear. It is crucial to discuss the release characteristics of nitrogen gas during the co-combustion of biomass and urban SS to address biomass fuel corrosion, agglomeration, and atmospheric pollution from nitrogen gas. The N conversion in the co-combustion of biomass and municipal SS was studied, showing that when the proportion of municipal SS changes from 0% to 30%, the reduction rate of NO and NO_x precursors is more significant. A mixture ratio of 70% urban sludge in corn stalk/sludge and cotton stalk/sludge is optimal for reducing NO and NO_x precursor release. Cotton stalk/sludge and corn stalk/sludge mixtures are superior to wheat straw/sludge mixtures. NO_x emissions decrease as air enters a higher phase and as the number of burn-out air locations increases. Additionally, flue gas circulation technology can improve the freeboard temperature, rapidly reduce the nitrogen and oxygen content, and further decrease NO_x emissions. Therefore, an auxiliary gas burner should be added when high freeboard temperature cannot be achieved solely through volatile combustion.

4.2.2. Pyrolysis

N emission reduction during pyrolysis can be viewed from two perspectives: (a) Controlling the emission of nitrogen gas compounds, and (b) recovering N resources in the pyrolysis products for reuse, thus reducing the release of N into the natural environment. During SS pyrolysis, as the temperature increases, proteins in the SS condense to form N-containing heterocyclic rings. The addition of $Ca(CH_3COO)_2$ has been found to increase N retention in SS chars and reduce the formation of N species in tars [128]. This effect is primarily due to the inhibition of protein-N decomposition and the promotion of stable nitrogen species formation in the char. Conversely, the addition of CH_3COONa enhances the decomposition of N species in the sludge, such as protein and inorganic N. The concentrations of $Ca(CH_3COO)_2$ and CH_3COONa in N-containing gas emissions are significantly reduced because acetone is produced when acetate is heated, and acetone easily reacts with NH to form binary clusters or amines [128]. The pyrolysis of proteins and amino acids in SS is influenced by their existing forms and the presence of other substances, such as carbohydrates [129]. To improve the pyrolysis efficiency of SS and better recover N resources, different pyrolysis schemes should be devised for SS with varying carbohydrate contents (mainly lignin, cellulose, and hemicellulose). The addition of CaO significantly affects the form and process of nitrogen emissions during SS pyrolysis at various temperatures, reducing the activation energy required for pyrolysis [130]. This facilitates the retention of nitrogen in the char and its conversion into the gas phase, while simultaneously hindering its migration to tar. Therefore, studying the effect of different proportions of lignin on nitrogen transport during sludge pyrolysis is useful for understanding the nitrogen transport mechanism during the pyrolysis process.

4.2.3. Gasification

During SS gasification, NO_x precursors are the primary nitrogen-containing gas pollutants, while the total yield of coke-N and tar-N is only 10 % [131]. The oxidative activity of oxygen carrier supports can significantly promote the oxidative conversion of NO_x precursors, tar nitrogen, and coke nitrogen to N_2. Among these, NiO-modified copper slag has been used as an oxygen carrier for chemical cycle gasification. This promotes the oxidation and catalytic cracking reactions of NO precursors, Char-N, and Tar-N, increasing the N yield from 52.49 % to 91.98 % [132]. The release of NH_3 and HCN is mainly concentrated in the first 5 min, and both NH_3-N and HCN–N gradually decrease with the number of cycles. Under the synergistic effect of Fenton's reagent and CaO regulator, the homogeneous phase reduction of NO_x precursors accounts for 50.59 %–98.76 % of the final reduction [133]. Additionally, in a heterogeneous process, iron catalysis converts nitrile-N and pyrrole-N in coke to more stable indole-N in tar [133]. Furthermore, both char-N and tar-N can be decomposed into N_2 in the presence of CaO. Currently, SS gasification is still in its emerging stage, and research on N emission reduction in this process is relatively limited. Future research should focus on in-depth studies of the N conversion process and emission reduction measures during SS gasification.

4.2.4. Hydrothermal processing

During sludge hydrothermal treatment, nitrogen is primarily decomposed into ammonia nitrogen (NH_4^+) and nitrate nitrogen in the liquid phase products. The residual nitrogen in solid products mainly exists as pyridine-N and pyrrole-N. A large number of nitrogen-containing compounds in biocrude primarily include heterocycle-N, amine-N, amide-N, and nitrile-N. As the reaction intensity increases, nitrogen in the solid gradually transfers into the biocrude and the water phase, significantly increasing the contents of nitrogen-containing heterocyclic compounds in the biocrude and

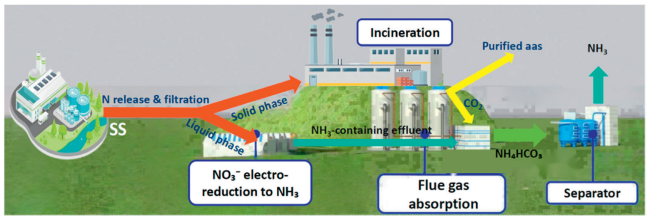


Fig. 3. Nitrate electroreduction to ammonia for N emission reduction strategy: schematic diagram of N recovery and CO_2 capture in SS Treatment.

NH_4^+ in the water phase [134]. Therefore, hydrothermal pretreatment of high-water-content sludge can improve dehydration and decompose unstable protein nitrogen in the sludge into ammonia nitrogen and other substances in the liquid phase [135]. This process is conducive to subsequent nitrogen recovery and emission reduction. The nitrogen emission problem during SS processing remains a significant challenge. Recovering nitrogen from SS before final disposal is expected to achieve environmental protection and resource recovery. Struvite (MAP) crystallization is considered a promising method for the simultaneous recovery of P and nitrogen from SS. After hydrothermal treatment at 200 °C for 30 min, the concentration of NH_3-N in the liquid phase was 456.85 mg-N/L. During the subsequent MAP precipitation, the optimal conditions for nitrogen recovery were determined to be pH 9 and a Mg/P molar ratio of 1 [136]. With an MAP purity of 84.24 %, the nitrogen recovery in the hydrolyzate was 54.88 % [136]. Further research on low-temperature SS treatment is needed to reduce the environmental impact of hydrothermal treatment. Kinetic studies of hydrothermal treatment will also provide a better understanding of the general reaction pathways and formation of hydrothermal treatment products. Before commercializing hydrothermal treatment, the techno-economic analysis of the process and energy calculations should be thoroughly evaluated.

4.3. Electroreduction of nitrate to ammonia for N emission reduction strategy

The reduction of N during S treatment faces several challenges. Electrochemical nitrate reduction (NITRR) not only efficiently removes nitrogen from SS but also converts it into NH_3, which is of significant importance for nitrogen recovery. By rationally designing electrocatalysts and optimizing reaction conditions, NITRR can markedly enhance nitrogen conversion efficiency during SS treatment and reduce environmental nitrogen pollution. Despite demonstrating excellent nitrogen conversion performance, practical applications of electrochemical NITRR still face several challenges, such as the cost of electrocatalytic reactions, catalyst stability, and the feasibility of large-scale implementation, which require further investigation and resolution [137]. Moreover, the application of electrochemical nitrogen reduction in SS treatment necessitates the development of economically viable and sustainable nitrogen removal and recovery processes. As shown in Fig. 3, based on the work of Fan *et al.* [137], we have further improved the electrochemical nitrate reduction-NH_3 technology, enabling not only CO_2 capture but also significantly enhancing nitrogen recovery efficiency in SS.

In the future, research on NITRR for nitrogen recovery will continue to progress in several directions: optimizing and innovating catalysts, further investigating novel electrocatalytic materials, and optimizing catalyst structure and performance to improve nitrogen reduction efficiency; optimizing reaction conditions, such as adjusting the electrolyte's pH and current density, to further enhance Faradaic efficiency and NH_3 yield; and developing electrochemical reactors and systems suitable for SS treatment, integrating them

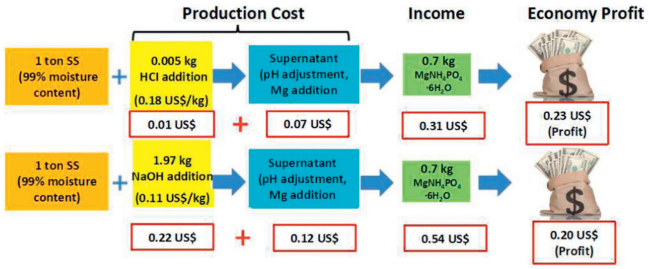


Fig. 4. MAP recovery cost calculation.

with existing SS treatment processes to achieve efficient nitrogen removal and recovery.

5. P recovery from SS

5.1. Physicochemical processes

5.1.1. Precipitation

Physicochemical methods are the most commonly used techniques to recover P in SS treatment, primarily including precipitation and adsorption. Struvite precipitation has been widely employed to recover phosphorus from SS. In addition to recycling P resources in SS, N elements can also be recovered. The formation of magnesium ammonium phosphate (MAP) requires the presence of specific concentrations of magnesium ions, ammonia nitrogen, and positive phosphate in the solution. The basic reaction principle can be expressed by the following chemical equations (Eqs. 1–3) [138]:

$$Mg^{2+} + NH_4^+ + PO_4^{3-} + 6H_2O = MgNH_4PO_4 \cdot 6H_2O \downarrow \quad (1)$$

$$Mg^{2+} + NH_4^+ + HPO_4^{2-} + 6H_2O = MgNH_4PO_4 \cdot 6H_2O \downarrow + H^+ \quad (2)$$

$$Mg^{2+} + NH_4^+ + H_2PO_4^- + 6H_2O = MgNH_4PO_4 \cdot 6H_2O \downarrow + 2H^+ \quad (3)$$

Although in theory Mg:N:P = 1:1:1 in the composition of MAP, other ions are inevitable in the actual sewage. To recover P as MAP from the liquid phase of hydrothermal carbonation-treated P-rich dairy sludge, a Mg:NH_4^+:P molar ratio of 1.73:1.14:1 was used for the salt dosage for struvite precipitation [139]. The P recovery was 99.96 %, and the P concentration in the remaining liquid phase was <2 mg/L [139]. Xu *et al.* (2022) used HCl and NaOH pretreatment to estimate the cost of recovery of struvite in SS, as shown in Fig. 4 [140]. The cost of producing MAP from SS (99 % water content) is 0.08 $/kg and 0.34 $/kg, respectively. MAP is priced at 0.45 $/kg. Considering MAP production (0.7 kg/kg and 1.2 kg/kg), total MAP product revenue is 0.31 $/kg and 0.54 $/kg. So the profits for recovering MAP from SS are 0.23 $/kg and 0.20 $/kg, respectively. Therefore, acid-base pretreatment combined with MAP recovery is economical. Besides MAP, the most commonly used form of phosphorus precipitation during P recovery is phosphate, the most common being calcium phosphate, aluminum phosphate, and iron phosphate. When the pH value is higher than 4, P co-precipitates with metals such as Al, Fe, and Ca in solution, but only Ca-P is the active ingredient of fertilizers [141]. With increasing pH, P mainly reacts with Al and Fe [142].

Precipitation methods like struvite formation are effective for P recovery from SS but face operational challenges due to the need for precise control over ion concentrations and pH levels, which fluctuate with wastewater composition. The efficiency of these methods is also limited by the unbalanced ratio of NH_3–N to phosphate typical in wastewater. Economic and environmental concerns, such as the cost of reagents and increased effluent salinity, further complicate these processes. In alkaline environments, Fe and Al salts offer alternatives but still require complex pH management and involve similar costs. Future improvements might include automated, sensor-based systems that dynamically adjust to SS variability, reducing chemical use and adapting to treatment challenges.

5.1.2. Adsorption

The adsorption method uses some porous or large specific surface area of solid substances to adsorb phosphate ions in water to achieve phosphorus removal and phosphorus recovery process [143–145]. In recent years, more and more scholars are devoted to the research of functional adsorbents to improve the recycling capacity of P in sludge. Activated carbon (AC) is one of the most common adsorbents in the adsorption process. The main reason for the adsorption is related to the content of Ca, Mg, Fe, Al, and other substances on the AC surface. Pore-like structures and some metal ions bound to AC can serve as adsorption sites for P [146,147]. Various cations on the AC surface combine with phosphate to form insolubles. Phosphate precipitation for phosphorus removal. Studies have shown that Fe and Al react more readily with phosphates under acidic conditions, whereas the reactions of Ca and Mg are more important under alkaline conditions. The fixation effect without the addition of MgO may be mainly reflected in the formation of Ca-P precipitates from Ca^{2+} and phosphate [148,149], similar to HAP. The process is as follows (Eqs. 4–6):

$$Ca^{2+} + 2H_2PO_4^- + 2H_2O = Ca(H_2PO_4)_2 \cdot H_2O \quad (4)$$

$$Ca(H_2PO_4)_2 + Ca^{2+} = 2CaHPO_4 + 2H^+ \quad (5)$$

$$2CaHPO_4 + Ca^{2+} = Ca_3(PO_4)_2 + 2H^+ \quad (6)$$

At present, the adsorbents for P recovery in SS are not limited to AC. The Mg/Ca-modified bagasse biochar (10 g/L) prepared by pyrolysis at 700 °C had a P adsorption capacity of 129.79 mg/g from the acid leaching solution of sludge ash [150]. In the process of biochar adsorption of P, chemical precipitation adsorption and pore filling are dominant. Compared with AC, biochar is cheaper and more economical. In order to further reduce the adsorption cost of P, biomass that itself contains metal salt ions can be selected. Eggshells (50 g/L) calcined at 900 °C for 30 min adsorbed 98 % of P from the H_2SO_4 leaching water of anaerobic sludge within 1 h, and part of the P was converted to calcite ($CaHPO_4{\cdot}2H_2O$) [151]. Adsorption, particularly using AC or biochar, is recognized for its simplicity and effectiveness in P recovery from SS. This method operates cleanly without secondary pollution [152,153] and can be cost-effective when biochar substitutes AC [154,155]. However, challenges arise in the regeneration and desorption of these adsorbents, which complicates operation and may lead to secondary pollution. Additionally, the sustainability of using large volumes of non-renewable adsorbents like AC needs further research. Developing more sustainable and easily regenerable adsorbents could greatly improve this method's overall viability and reduce operational difficulties.

5.2. Thermochemical processes

5.2.1. Recovery of P in dry thermochemical sludge ash

The thermochemical method for recovering P from sludge ash involves single incineration with distilled water in a rotary kiln, using additives to form a P-containing mineral phase with high solubility and low metal content. The two main objectives are phosphorus product recovery and heavy metal removal. During the phase transformation, components of the sludge ash (except

quartz and hematite) undergo a decomposition-recrystallization cycle, with the P-containing solid phase forming as old mineral phases disappear [156]. The fate of heavy metals during thermochemical processes is largely determined by their boiling points. Elements such as Al, Mg, Mn, K, and Si typically remain in the solid ash fraction, whereas more volatile elements such As, Pb, and Sn tend to vaporize and subsequently condense onto the surfaces of fly ash particles. Metal stability in their oxides or chlorides also influences this distribution.

To improve P recovery, heavy metal removal is crucial. Additives such as $CaCl_2$, $MgCl_2$, HCl, and gaseous HCl are used. Calcium chloride, for instance, reacts with gray manganese ore to form chloroapatite, transferring heavy metals from the solid phase to the gas phase [157]. $MgCl_2$ leads to bioavailable P fertilizers under acidic conditions [158]. At high temperatures (1000 °C), Mg and Ca react to form apatite, highly useful in energy plants [159]. Gaseous HCl is advantageous because it doesn't require mixing with ISSA before incineration, directly reacting with metals [19,157]. Although this method retains nearly all P in sludge ash, its high cost hinders broader adoption. A more economical approach involves co-incinerating sludge ash with chlorine-containing waste, like polyvinyl chloride.

5.2.2. Recovery of P in wet thermochemical sludge ash

As mentioned in the previous section, the recovery of P by thermochemical methods primarily involves recovering phosphorus products and removing heavy metals. The enrichment of P in the hydrothermal carbonation process is influenced by the medium, time, and temperature of the reaction. An acidic reaction medium can better promote the conversion of apatite phosphorus to non-apatite inorganic phosphorus and organic phosphorus to inorganic phosphorus, which is more conducive to phosphorus recovery [160]. During hydrothermal carbonation, most of the P ends up in the hydrochar as non-apatite inorganic phosphorus. In acidic media, the flocs in SS are broken down, hydrolyzing the organic matter and promoting the conversion of organic P to inorganic P [1,161]. The main acids currently studied and used are sulfuric acid [162] and citric acid [163]. Up to 90 % of phosphorus in hydrochar can be recovered at 220 °C by H_2SO_4 leaching [162]. The advantage of using citrate leaching is that, compared to other acids, it does not compete with MAP to form other phosphate precipitates, thereby reducing P losses [163]. The main disadvantage of citric acid is its high cost. In the hydrothermal carbonation process, some organic acids are also present in the liquid, which could potentially substitute for citric acid.

The effect of temperature on the migration of heavy metals during hydrothermal carbonation is similar to that of hydrothermal liquefaction. At a certain temperature, heavy metals tend to move to the liquid phase, which is related to the volatility of various heavy metals [164]. During hydrothermal carbonation, the behavior of various metals is roughly the same as in the pyrolysis process. Pyrolysis temperature has little effect on the evaporation of heavy metals during hydrothermal carbonation [165]. When the temperature increases, it promotes the transfer of heavy metals from the solid residue to the liquid phase. Hydrothermal carbonation can facilitate the partial conversion of heavy metals into relatively stable forms, promoting their fixation in hydrochar and reducing their ecotoxicity [166]. The recovery and reuse of phosphorus in sludge using hydrothermal carbonation are illustrated in Fig. 5. While the wet thermochemical process is effective in leaching P, it can also mobilize heavy metals, posing risks to both the environment and human health if not adequately managed. The process's efficiency is highly dependent on the reaction conditions, which require strict monitoring and control. Future research could focus on optimizing these conditions to maximize phosphorus recovery while minimizing heavy metal leaching.

5.3. Electrochemical techniques

Capacitive deionization (CDI), electrocoagulation (EC), and electroinduced precipitation (EIP) are notable electrochemical techniques for phosphorus recovery, each presenting unique challenges and opportunities for improvement. CDI is energy-efficient and capable of selective ion recovery, but its effectiveness is hindered by competitive ions like sulfate, necessitating the development of more selective membranes and effective pre-treatment steps. EC, while simple and effective, relies heavily on metal electrodes that can dissolve into the sludge, raising sustainability and cost concerns. Exploring alternative materials or recycling strategies for electrodes could address these issues. EIP, although highly effective under controlled conditions, requires significant energy and may struggle with variability in real-world wastewater settings. Enhancing energy efficiency and integrating renewable energy sources could improve EIP's sustainability and field adaptability.

5.3.1. Capacitive deionization

CDI is an emerging electrochemical technique for ion separation, where anions are adsorbed on the positive electrode and cations on the negative electrode during charging. During discharge, ions are released back to the SS source. The choice of electrode material is critical, as it affects the electrosorption capacity and efficiency. In SS resource recovery, CDI separates and concentrates P ions without altering the oxidation state of P. The separation is influenced by pH, as phosphate speciation is pH-dependent. In CDI systems, the diffusion rates of $H_2PO_4^{2-}$ and HPO_4^{2-} are slower, with HPO_4^{2-} ions having a higher energy advantage and preferentially storing in the electric double layer after prolonged charging [167].

A promising strategy for selective P recovery is using phosphate-selective anion-exchange membranes in membrane capacitive deionization systems [168]. However, sulfate in SS may interfere with phosphate separation [167]. CDI systems can also concurrently recover N and P. For SS containing heavy metals, a flow-electrode capacitive deionization (FCDI) system can remove heavy metals while recovering N and P nutrients [169], as shown in Fig. 6a. Phosphate recovery efficiency in FCDI is sensitive to electrode loading, as removal depends on physisorption and electrosorption. CDI is a promising P recycling method that requires no additional reagents, and its adsorption capacity, kinetics, and selectivity can be enhanced by optimizing electrode materials and electrochemical reactors. Further research to optimize CDI for P recovery is encouraged.

5.3.2. Electrocoagulation

EC is a conventional chemical solidification process utilizing soluble metal cations, typically aluminum or iron. During the EC process, Fe^{2+} and Al^{3+} ions are generated by anodic dissolution, while H_2 is produced at the cathode, as illustrated in Fig. 6b. Numerous studies have employed aluminum and iron anodes to recover P with recovery efficiencies exceeding 90 % [170]. The process of recovering P using the EC method involves the electrolytic dissolution of the anode material, generating Al^{3+} and Fe^{2+} ions that react directly with PO_4^{3-} in an acidic environment, leading to the precipitation of phosphate. This reaction is depicted in Eqs. 7 and 8:

$$3Fe^{2+} + 2PO_4^{3-} = Fe_3(PO_4)_2 \quad (7)$$

$$Al^{3+} + PO_4^{3-} = AlPO_4 \quad (8)$$

The choice of electrodes directly affects the efficiency of P recovery in EC. Al and Fe electrodes in a monopolar configuration individually recovered 99.3 % and 80.7 % of P, respectively [171].

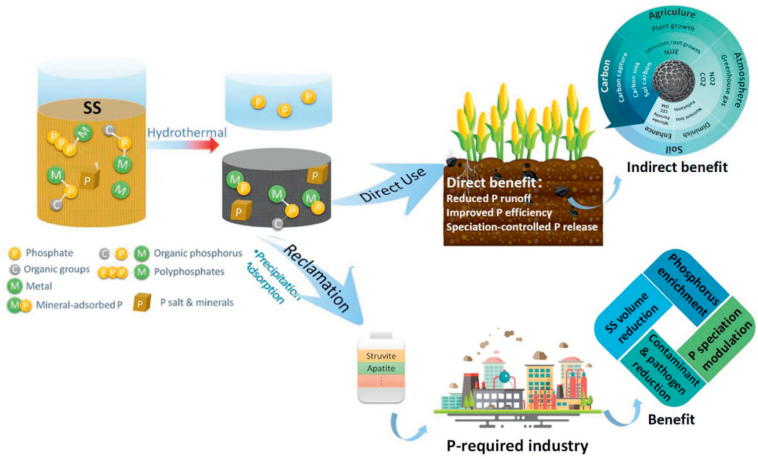


Fig. 5. Hydrochar recovery and reuse of P in SS.

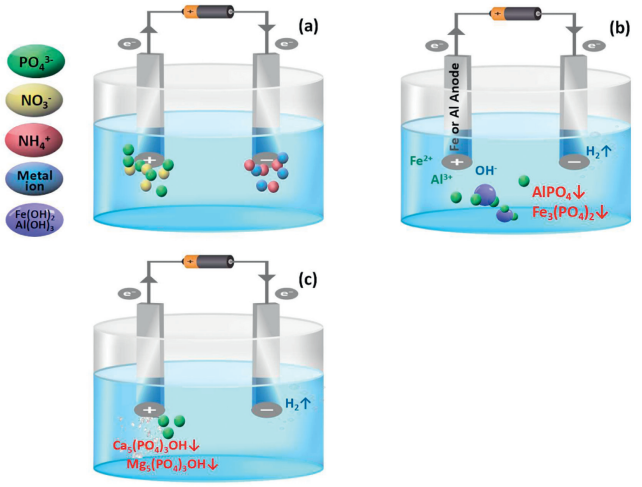


Fig. 6. Electrochemical methods for phosphorus recovery from SS: (a) Capacitive deionization, (b) electrocoagulation and (c) electroinduced precipitation.

This phenomenon may be due to the higher adsorption capacity of $Al(OH)_3$ for phosphate compared to $Fe(OH)_2$. Further studies found that Fe electrodes removed phosphates only by adsorption into metal hydroxides, whereas Al electrodes could remove phosphates by both precipitation and adsorption into metal hydroxides [172]. For this reason, Al electrodes are widely used in EC. EC technology is environmentally friendly, with easy operation, simple equipment, and no need for added chemicals. Compared to other P recovery methods, EC is characterized by rapid sedimentation. However, metal anodes represent the main operating cost of EC. Many metalworking and machining workshops worldwide generate large amounts of metal scrap that can be used as electrodes for the EC process. Using scrap metal as electrodes benefits the environment and reduces the costs associated with EC.

5.3.3. Electroinduced precipitation

EIP is a promising method to recover P [173]. The formation of P mineral precipitation in EIP is achieved by increasing the pH at the cathode surface, while phosphate ions do not directly participate in the electrochemical reaction, as shown in Fig. 6c. At the cathode, water molecules are reduced to H_2 and OH^-, resulting in a local pH increase (Eq. 9). Simultaneously, due to electromigration, cations (*e.g.*, Mg^{2+}/Ca^{2+}) move toward the cathode and accumulate near it. Consequently, the saturation index of P minerals increases, driving the formation and precipitation of P minerals on the cathode (Eq. 10). At the anode, water oxidation produces H^+, which neutralizes the OH^- produced at the cathode (Eq. 11). Therefore, while the local pH at the cathode increases, the pH of the bulk solution is not expected to change significantly in the electrochemical system [173,174].

$$\text{Cathode}: 4H_2O + 4e^- \rightarrow 4OH^- + 2H_2\uparrow \quad (9)$$

$$5Ca^{2+}/Mg^{2+} + 3HPO_4^{2-} + 4OH^- \rightarrow Ca/Mg_5(PO_4)_3OH\downarrow + 3H_2O \quad (10)$$

$$\text{Anode}: 2H_2O \rightarrow 4H^+ + O_2\uparrow + 4e^- \quad (11)$$

As described in Section 5.1.1, MAP precipitation is a well-established method for recovering P from SS. Due to the low concentration of Mg^{2+} in SS, the addition of magnesium salts such as $MgCl_2$ is required to induce MAP precipitation [175]. In EIP, MAP precipitation can provide dissolved Mg^{2+} through sacrificial magnesium electrodes. Magnesium electrodes undergo rapid anodic dissolution, leading to high precipitation rates, and non-galvanic corrosion reduces the energy consumption of the process [176]. When the Mg content in SS is low, recovering P as calcium phosphate (Ca-P) is more suitable [177]. Since Ca^{2+} is widely present in SS, no additional dosing is required during the EIP process for the recovery of Ca-P, especially in calcium-rich dairy industry SS. Furthermore, and more importantly, EIP recovers bioavailable P in one process with little or no post-processing. Therefore, EIP can help establish and maintain a new and sustainable P cycle by utilizing the P accumulated in SS as a substitute for mined P rock. In electrochemical systems, the recovery and utilization of by-products such as H_2 can reduce the overall energy cost of P recovery. A dual benefit can be obtained by recycling P from SS and simultaneously producing H_2. This can be achieved by optimizing system design, such as electrode spacing, electrode materials, and reactor configuration. The related research and development of antifouling films and electrode materials should be encouraged to advance the field of electrochemistry.

6. Understand the impact of limiting factors on N and P resource recovery

Efficient recycling of N and P from SS, while minimizing environmental risks, is a significant challenge due to SS complexity. Developing methods to yield high-quality N and P products, while reducing energy consumption and costs, is essential for addressing both technical and environmental concerns. Unlike municipal

SS, industrial SS properties are plant-specific. For example, pharmaceutical SS, with high organic pollutants, is best treated with ozone oxidation, which can release N and P and remove pathogenic microorganisms without secondary pollution [178]. Steelmaking and electroplating SS, which contain heavy metals, pose risks during composting by promoting microbial community changes, leading to increased diversity but also secondary pollution from metal migration [179,180]. Composting should be avoided for these types\break of SS.

Local environmental and economic conditions should also be considered. In areas with low SS production, mobile heat treatment methods (pyrolysis, gasification) and *in-situ* composting technologies are more cost-effective. Pyrolysis reduces SS weight, and products can be treated for N and P recovery. *In-situ* composting is more suitable for reducing polluting gases and N loss due to SS's high water content. In large cities, where SS production is concentrated, incineration and large-scale composting are preferable, while colder regions benefit from freeze-thaw techniques. Energy-intensive treatments, such as ultrasonic, ozone, and electrochemical methods, are less suitable in economically underdeveloped areas due to high costs.

Despite higher energy consumption and membrane replacement costs, electrochemical methods produce purer P products compared to precipitation, crystallization, and adsorption. In electrochemical processes, anion exchange membranes selectively allow PO_4^{3-}, HPO_4^{2-}, and $H_2PO_4^-$ ions to pass through, avoiding impurities, resulting in high-purity P products. During SS gasification and pyrolysis, nitrogen is converted into NO_x, causing secondary pollution like photochemical smog. For urban areas, hydrothermal carbonization, with lower toxicity, should be prioritized.

7. Future prospects and challenges

Current N and P recycling methods face significant technical and environmental challenges, highlighting the need for extensive research and enhancement in SS management. A tailored, systematic approach is necessary to select optimal treatment methods for each scenario, aiming to improve yield. Future efforts should prioritize lowering environmental risks, reducing energy use, and cutting costs in recovery processes, with research focused on more controlled and efficient SS resource recovery methods. Key areas of focus include:

(1) Synergistic combinations of techniques: Exploring the integration of multiple technologies offers promising potential to reduce nitrogenous pollutant emissions during SS treatment. For instance, combining biological nitrogen removal, chemical treatments, and advanced oxidation processes can enhance treatment efficiency and minimize environmental impacts. Future research should focus on optimizing these integrated approaches to achieve more efficient, sustainable, and energy-saving treatment processes.
(2) Cost-effective sources of calcium and magnesium: The use of calcium and magnesium is essential for the formation of phosphorus-rich crystals or precipitates during phosphorus recovery. However, the current sources of these minerals are both expensive and unsustainable. Future research should explore alternative, cost-effective, and sustainable sources of calcium and magnesium, such as waste streams or by-products from other industrial processes. This would not only make the recovery process more economically viable but also promote circular economy principles in wastewater treatment.
(3) Comprehensive economic and environmental assessments: To ensure the widespread adoption of SS recycling methods, it is crucial to conduct thorough economic and environmental assessments. These assessments should evaluate the lifecycle costs, energy consumption, carbon emissions, and potential environmental risks associated with various recovery techniques. Such evaluations will provide a clearer understanding of the trade-offs between technologies and offer actionable insights for stakeholders, enabling them to make informed decisions when investing in and implementing SS recycling solutions.

8. Conclusion

Effective SS management, crucial for carbon neutrality, encompasses nitrogen emission reduction and phosphorus resource recovery. Advanced technologies such as ultrasonic treatment, aerobic/anaerobic digestion, and thermochemical conversion have significantly improved nitrogen and phosphorus recovery efficiencies. Notably, thermochemical conversion enhances the transformation of N and P in sludge compared to traditional methods, increasing nitrogen recovery by approximately 30 % and boosting phosphorus recovery to over 85 %. Innovative strategies, including bio-augmented phosphorus recovery and electrochemical techniques, have successfully reduced nitrogen emissions and enhanced phosphorus resource utilization. These strategies not only improve the environmental sustainability of sludge treatment but also significantly reduce costs; for instance, electrochemical technology has decreased phosphorus recovery expenses by about 20 % while controlling nitrogen emissions. Overall, this study provides a range of effective techniques and strategies for sludge resource recovery, offering crucial theoretical and practical insights for sustainable management and recycling of nitrogen and phosphorus during sludge treatment processes. This review holds significant practical importance and broad application prospects for advancing N and P recovery in the sludge treatment industry.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

Lu Liu: Writing – original draft, Conceptualization. **Yihe Wang:** Writing – original draft, Data curation. **Jialin Wang:** Writing – original draft, Investigation. **Chao He:** Writing – review & editing. **Jun Zhang:** Writing – review & editing, Funding acquisition.

Acknowledgments

This study was supported by the National Key R&D Program of China (No. 2023YFC3902800), the National Natural Science Foundation of China (No. 52570153), the State Key Laboratory of Urban Water Resource and Environment, HIT (No. 2023DX11), the Research Project of Daqing Oilfield in Heilongjiang Province (25DQYTSG024-07-03). The authors also appreciate the Science Foundation of National Engineering Research Center for Safe Disposal and Resources Recovery of Sludge (No. Z2024A014). We are grateful for the constructive comments and suggestions from Prof. Yan Zhou, Nanyang Technological University, Singapore, during the initial draft preparation of this work.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.cclet.2025.111431.

References

[1] L. Wang, Y. Chang, A. Li, Renew. Sustain. Energ. Rev. 108 (2019) 423–440.
[2] E. Pozzebon, L. Seifert, Environ. Health 22 (2023) 57.
[3] S. Chu, D. Wu, L. Liang, et al., Sci. Rep. 7 (2017) 13408.
[4] J. Zhao, T. Hou, Z. Lei, K. Shimizu, Z. Zhang, Sci. Total Environ. 783 (2021) 146966.
[5] R. Kumar, T. Dada, A. Whelan, et al., J. Hazard. Mater. 452 (2023) 131212.
[6] S. Cho, Y. Kim, M. Kim, et al., Process Safety Environ. Prot. 162 (2022) 813–824.
[7] S. Guo, T. Liu, D. Che, H. Liu, B. Sun, Environ. Technol. Innov. 23 (2021) 101594.
[8] S. Solarin, V. Yilanci, M. Gorus, J. Clean. Prod. 324 (2021) 129298.
[9] M. Dudziak, S. Werle, A. Marszałek, S. Sobek, Fuel 318 (2022) 123678.
[10] J. Jia, L. Zhao, Z. Liu, et al., Energy 242 (2022) 122534.
[11] Q. Yang, Q. Yao, D. Ma, et al., J. Anal. Appl. Pyrolysis 163 (2022) 105486.
[12] Z. Chen, Y. Li, Y. Peng, et al., Sci. Total Environ. 825 (2022) 154047.
[13] X. Zhang, J. Zhao, L. Ding, et al., Ecol. Indic. 138 (2022) 108821.
[14] K. Hui, B. Kou, Y. Jiang, et al., Sci. Total Environ. 811 (2022) 151420.
[15] X. Li, H. Jiang, L. Zhu, et al., Chemosphere 351 (2024) 141181.
[16] H. Jiang, Y.J. Dai, Chemosphere 311 (2023) 136884.
[17] K. Kor, H. Ershadifar, A. Ghazilou, E. Koochaknejad, Mar. Pollut. Bull. 173 (2021) 113125.
[18] X. Liu, Z. Shao, Y. Wang, et al., Environ. Res. 216 (2023) 114651.
[19] M. Zin, D. Tiwari, D. Kim, J. Ind. Eng. Chem. 86 (2020) 136–143.
[20] L.L. Kong, Q. Shan, Y.L. Lai, et al., J. Soils Sediments 24 (2024) 2638–2651.
[21] D. Cecconet, A.G. Capodaglio, Sustainability 14 (2022) 14841.
[22] S. Hoang, N. Bolan, A. Madhubashani, et al., Environ. Pollut. 293 (2022) 118564.
[23] Y. Huang, P. Chiueh, S. Lo, et al., Energy Procedia 158 (2019) 67–72.
[24] M. Koyama, A. Kakiuchi, K. Nakasaki, Sci. Total Environ. 802 (2022) 149961.
[25] J. Alonso, A. de Abreu, C. Andreoli, et al., Environ. Monit. Assess. 196 (2024) 34.
[26] K. Pei, K. Xiao, H. Hou, et al., Environ. Res. 191 (2020) 110050.
[27] S. Guo, T. Liu, J. Hui, et al., Energy 189 (2020) 116217.
[28] S. Chen, X. Dai, D. Yang, B. Dong, J. Environ. Sci. 105 (2021) 11–21.
[29] Y.F. Li, C. Hong, Y.J. Wang, et al., ACS Sustain. Chem. Eng. 8 (2020) 7721–7740.
[30] J. Tang, L. Zhuang, Z. Yu, et al., Sci. Total Environ. 656 (2019) 29–38.
[31] C. Zhang, Y. Xu, M. Zhao, H. Rong, K. Zhang, J. Hazard. Mater. 344 (2018) 163–168.
[32] M. Hossain, V. Strezov, K. Chan, A. Ziolkowski, P. Nelson, J. Environ. Manag. 92 (2011) 223–228.
[33] T.C. Sichler, C. Adam, D. Montag, M. Barjenbruch, J. Clean. Prod. 332 (2022) 130130.
[34] H. Kominko, K. Gorazda, Z. Wzorek, J. Environ. Manag. 301 (2022) 114417.
[35] Z. Bairq, R. Li, Y. Li, et al., J. Clean. Prod. 190 (2018) 339–349.
[36] M. Munir, B. Li, I. Mardon, et al., J. Clean. Prod. 232 (2019) 1043–1052.
[37] B. Yu, J. Luo, H. Xie, et al., Sci. Total Environ. 786 (2021) 147437.
[38] B. Yu, X. Xiao, J. Wang, et al., Bioresour. Technol. 341 (2021) 125899.
[39] L. Staal, A. Petersen, C. Jorgensen, et al., Water Res. 157 (2019) 346–355.
[40] X. Chen, Y. Zhao, C. Zeng, et al., Bioresour. Technol. 289 (2019) 121657.
[41] M. Li, Y. Tang, X. Lu, Z. Zhang, Y. Cao, Water Res. 140 (2018) 90–99.
[42] L. Yang, X. Guo, S. Liang, et al., Water Res. 233 (2023) 119769.
[43] Z. Zhang, S. Baroutian, M. Munir, B. Young, Bioresour. Technol. 245 (2017) 234–241.
[44] C. Phalakornkule, S. Nuchdang, S. Vatanyoopaisarn, J. Biosci. Bioeng. 123 (2017) 474–481.
[45] A. Montusiewicz, M. Lebiocka, A. Rozej, E. Zacharska, L. Pawlowski, Bioresour. Technol. 101 (2010) 3466–3473.
[46] J. Gao, Y. Wang, Z. Li, J. Clean. Prod. 295 (2021) 126288.
[47] R. Zhang, Y. Mao, L. Meng, Sep. Purif. Technol. 276 (2021) 119359.
[48] C. Gong, J. Jiang, D. Li, Sci. Total Environ. 532 (2015) 495–500.
[49] U. Ushani, J. Banu, K. Tamilarasan, et al., Bioresour. Technol. 241 (2017) 710–719.
[50] D. Cao, F. Tian, X. Wang, et al., Chemosphere 283 (2021) 131181.
[51] W. Li, N. Yu, Q. Liu, et al., Sci. Total Environ. 635 (2018) 699–704.
[52] Y. Lin, N. Zheng, H. Wang, Renew. Sustain. Energy Rev. 155 (2022) 111857.
[53] L. Liu, J. Zhang, Y. Chen, et al., Sustainability 15 (2023) 7698.
[54] National Economic and Social Development Statistical Bulletin, 2021. Available online: https://www.stats.gov.cn/zt_18555/zthd/lhfw/2022/lh_hgjj/202302/t20230214_1903465.html (accessed on 4 November 2022).
[55] W. Zhang, Y. Jiang, X. Xie, Carbon Neutral. 2 (2023) 603–615.
[56] X. Sun, B. Liu, L. Zhang, et al., Sci. Total Environ. 807 (2022) 150773.
[57] A. Chiavola, C. Salvati, S. Bongirolami, C. Di Marcantonio, M. Boni, J. Water Process Eng. 42 (2021) 102114.
[58] W. Mu, W. Dagnew, Case Stud, Chem. Environ. Eng. 5 (2022) 100178.
[59] D. Ge, S. Huang, J. Cheng, et al., J. Clean. Prod. 359 (2022) 132049.
[60] C. Wang, W. Wei, X. Dai, B. Ni, Chem. Eng. J. 433 (2022) 134459.
[61] D. Ikumi, T. Harding, Water Res. 170 (2020) 115333.
[62] J. Wang, H. Mao, J. Zhou, C. Yao, Y. Wang, Processes 11 (2023) 3206.
[63] Y. Chen, H. Lin, N. Shen, Bioresour. Technol. 294 (2019) 122160.
[64] R. Castellanos, M. Dezotti, J. Bassin, Chem. Eng. J. 176 (2021) 108221.
[65] A. Cosenza, D. Di Trapani, P. Bosco Mofatto, G. Mannina, Chemosphere 347 (2024) 140695.
[66] G. Cheng, Z. Li, L. Sun, Y. Li, J. Fu, Process Saf. Environ. Prot. 138 (2020) 148–156.
[67] P. Zhou, D. Li, C. Zhang, et al., Sci. Total Environ. 921 (2024) 171175.
[68] L. Mu, L. Zhang, K. Zhu, et al., Sci. Total Environ. 704 (2020) 135429.
[69] H. Xu, Y. Liu, B. Yang, et al., Chem. Eng. J. 405 (2021) 126506.
[70] H. Xu, L. Che, Y. Liu, et al., Water Res. 199 (2021) 117197.
[71] H. Durak, Processes 11 (2023) 2092.
[72] Z. Mei, D. Chen, J. Zhang, et al., Waste Manage. 106 (2020) 77–87.
[73] G. Jiang, D. Xu, B. Hao, et al., J. Cleaner Prod. 311 (2021) 127811.
[74] X. Niu, L. Shen, Chem. Eng. J. 335 (2018) 621–630.
[75] H. Nan, Z. Xiao, L. Zhao, et al., ACS Sustain. Chem. Eng. 8 (2020) 12197–12207.
[76] G. Qi, C. Li, Y. Mei, et al., J. Environ. Chem. Eng. 8 (2020) 104452.
[77] Y. Tian, J. Zhang, D. Wu, Z. Li, Y. Cui, Water Res. 47 (2013) 279–288.
[78] X. Meng, Q. Huang, J. Xu, Waste Disposal Sustain. Energy 2 (2020) 79.
[79] P.D. Solanki, M. Oza, H.O. Jethwa, M.J. Joshi, J. Mater. Sci. Mater. Electron. 35 (2024) 1141.
[80] Y.H. Chen, T.N.L.T. Ngo, K.Y. Chiang, Int. J. Hydrogen Energy 46 (2021) 14083–14095.
[81] H.Y. Chen, R. Shan, S. Li, et al., Sci. Total Environ. 913 (2024) 169535.
[82] T. Hannl, G. Haggstrom, A. Hedayati, et al., Fuel Process. Technol. 227 (2022) 107102.
[83] A. Hedayati, H. Sefidari, C. Boman, et al., Fuel Process. Technol. 217 (2021) 106805.
[84] C. Lin, Z. Xin, S. Yuan, J. Sun, B. Dong, Z. Xu, Water Res. 251 (2024) 121133.
[85] M. Marzbali, S. Kundu, P. Halder, et al., Chemosphere 279 (2021) 130557.
[86] W. Chen, M. Haque, T. Lu, A. Aierzhati, G. Reimonn, Curr. Opin. Environ. Sci. Health 14 (2020) 63–73.
[87] Centrisys-CNP-Logo-3c-stacked. https://www.centrisys-cnp.com/pondus (accessed 9 April 2024).
[88] R. Huang, Y. Tang, L. Luo, Waste Manage. 121 (2021) 276–285.
[89] K. Rathika, S. Kumar, B. Yadav, Sci. Total Environ. 906 (2024) 167828.
[90] K. Yang, J. Sun, H. Liu, W. Yang, L. Dong, Polymers (Basel) 15 (2023) 2396.
[91] R.P. Ipiales, G. Lelli, E. Diaz, et al., Environ. Res. 246 (2024) 118098.
[92] P. Das, S. Khan, M. AbdulQuadir, et al., Sci. Total Environ. 715 (2020) 136775.
[93] L.M. Liu, Y.B. Zhai, H.X. Wang, et al., Waste Manage. 158 (2023) 164–175.
[94] Y. Zhang, S. Zhang, H. Li, C. Wang, F. Jiang, J. Lyu, Chemosphere 257 (2020) 127140.
[95] H. Liu, Q. Zhang, H. Hu, P. Liu, X. Hu, A. Li, H. Yao, Proc. Combust. Inst. 35 (2015) 2759–2766.
[96] S. Liang, H. Chen, X. Zeng, et al., Water Res. 159 (2019) 242–251.
[97] J. Pan, H. Cai, Z. Zhang, et al., Bioresour. Technol. 270 (2018) 467–475.
[98] M.U. Hassan, G.Q. Huang, R. Munir, T.A. Khan, M.A. Noor, Agronomy-Basel 14 (2024) 1583.
[99] W. Li, C. Wu, K. Wang, L. Meng, L. Lv, Int. Biodeterior. Biodegr. 124 (2017) 297–303.
[100] S. Li, C. Wang, Z. Luo, X. Zhu, Energy Fuels 34 (2020) 12654–12664.
[101] A. Nagy, T. Magyar, N. Kiss, J. Tamás, int. J. Phytoremediat. 25 (2023) 1510–1523.
[102] E. Nie, D. Gao, G. Zheng, Bioresour. Technol. 315 (2020) 123812.
[103] Y. Zhu, Y. Zhai, S. Li, et al., Chemosphere 291 (2022) 133053.
[104] M. Awasthi, Q. Wang, H. Huang, et al., Bioresour. Technol. 216 (2016) 172–181.
[105] M. Awasthi, Q. Wang, X. Ren, et al., Bioresour. Technol. 219 (2016) 270–280.
[106] J. Liang, Y. Shen, Z. Shou, H. Yuan, X. Dai, N. Zhu, Bioresour. Technol. 264 (2018) 116–122.
[107] H. Huang, J. Liu, L. Ding, J. Clean. Prod. 102 (2015) 437–446.
[108] A. Petrovic, M. Simonic, L. Cucek, J. Environ. Manage. 290 (2021) 112593.
[109] Y. Gao, L. Tan, F. Liu, et al., Bioresour. Technol. 341 (2021) 125746.
[110] S. Xu, L. Li, X. Guo, Bioresour. Technol. 341 (2022) 125746.
[111] M. Seleiman, A. Santanen, P. Makela, Resour. Conserv. Recy 155 (2020) 104647.
[112] S. Zhou, Y. Li, Z. Jiang, J. Cleaner Prod. 339 (2022) 130759.
[113] L. Cao, L. Liao, R. Li, Sci. Total Environ. 773 (2021) 145561.
[114] Y. Liu, L. Ding, B. Wang, Q. He, D. Wan, Bioresour. Technol. 309 (2020) 123357.
[115] L. Meng, W. Li, X. Zhang, et al., Heliyon 6 (2020) e04988.
[116] M. Toledo, M. Gutierrez, J. Siles, et al., J. Cleaner Prod. 210 (2019) 1098–1108.
[117] M. Toledo, P. Marquez, J. Siles, et al., J. Environ. Manage. 247 (2019) 205–213.
[118] K. Tsigkou, D. Zagklis, P. Tsafrakidou, et al., Waste Manage. 118 (2020) 655–666.
[119] I. Ezemagu, M. Ejimofor, M. Menkitia, C. Diyoke, Environ. Challenges 5 (2021) 100288.
[120] R. Serna-Garcia, P. Ruiz-Barriga, A. Bouzas, J. Environ. Manage. 281 (2021) 111890.
[121] P. Cardoso, P. Gonçalves, G. Alves, R. Pegoraro, R. Sampaio, Rev. Ceres 68 (2021) 471–483.
[122] P. Cardoso, P. Gonsalves, G. Alves, et al., J. Environ. Manage. 314 (2022) 115076.
[123] L. Meng, W. Li, S. Zhang, et al., Sci. Rep. 11 (2021) 1319.
[124] Y. Zhao, W. Li, L. Chen, et al., Bioresour. Technol. 311 (2020) 123461.
[125] Y. Tang, B. Dong, X. Dai, Chem. Eng. J. 429 (2022) 132247.
[126] J.P. Xiong, Y. Su, H.W. Qu, et al., Sci. Total Environ. 864 (2023) 161065.
[127] W.J. Yang, Z.W. He, Y.X. Ren, et al., Bioresour. Technol. 358 (2022) 127430.
[128] S. Cheng, Y. Qiao, J. Huang, et al., Proc. Combust. Inst. 37 (2019) 2715–2722.
[129] S.S. Chen, B. Dong, X.H. Dai, et al., Waste Manage. 88 (2019) 309–318.
[130] J. Meng, J. Wang, F. Yang, F. Cheng, Sci. Rep. 458 (2023) 141039.
[131] S. Fang, Z. Deng, Y. Lin, et al., Energy 228 (2021) 120448.
[132] S. Fang, Z. Deng, Y. Lin, et al., Energy 216 (2021) 119247.
[133] G. Lu, H. Liu, Q. Zhang, et al., Fuel 241 (2019) 1109–1116.
[134] B. Hao, W. Yang, Y. Wang, et al., J. Anal. Appl. Pyrolysis 165 (2022) 105562.

[135] L. Chen, W. Feng, J. Fan, K. Zhang, Z. Gu, Sci. Total Environ. 711 (2020) 135155.
[136] Y. Yu, Z. Lei, T. Yuan, et al., Bioresour. Technol. 243 (2017) 634–640.
[137] K. Fan, W.F. Xie, J.Z. Li, et al., Nat. Commun. 13 (2022) 7958.
[138] H. Liu, G. Hu, I. Basar, et al., Chem. Eng. J. 417 (2021) 129300.
[139] C. Numviyimana, J. Warchol, N. Khalaf, J. Leahy, K. Chojnacka, J. Environ. Chem. Eng. 10 (2022) 106947.
[140] H. Xu, L. Guo, M. Gao, et al., Sci. Total Environ. 825 (2022) 154110.
[141] L. Fang, J. Li, S. Donatello, et al., Chem. Eng. J. 348 (2018) 74–83.
[142] X. Zhu, B. Liu, L. Sun, et al., Bioresour. Technol. 369 (2023) 128454.
[143] Y. Dai, W. Wang, L. Lu, L. Yan, D. Yu, J. Clean. Prod. 257 (2020) 120573.
[144] L. Liu, Y. Dai, Environ. Sci. Pollut. Res. 28 (2021) 48379–48391.
[145] L. Liu, W. Fang, M. Yuan, et al., J. Environ. Chem. Eng. 9 (2021) 106238.
[146] H. Jiang, X. Li, J. Bai, et al., Chem. Eng. Sci. 283 (2024) 119403.
[147] S. Ye, G. Zeng, H. Wu, et al., Resour. Conserv. Recycl. 140 (2019) 278–285.
[148] S. Ren, S. Wang, Y. Liu, et al., Sep. Sci. Technol. 58 (2023) 2578–2602.
[149] Y. Dai, Y. Liu, Y. Wang, et al., Pol. J. Environ. Stud. 32 (2023) 1479–1489.
[150] L. Fang, J. Li, S. Donatello, et al., J. Clean. Prod. 244 (2020) 118853.
[151] E. Panagiotou, N. Kafa, L. Koutsokeras, et al., J. Clean. Prod. 178 (2018) 419–428.
[152] L. Liu, X. Li, X. Wang, et al., Environ. Pollut. 308 (2022) 119610.
[153] L. Liu, X. Wang, W. Fang, et al., Environ. Res. 204 (2022) 111919.
[154] L. Liu, Z. Liu, S. Wang, et al., Environ. Eng. Manag. J. 21 (2022) 1815–1824.
[155] L. Liu, M. Yuan, X. Wang, et al., Environ. Technol. Innov. 24 (2022) 101841.
[156] T. Fournie, T. Rashwan, C. Switzer, J. Gerhard, Waste Manage. 137 (2022) 241–252.
[157] L.F. Lin, X.Z. Cao, J.F. Xi, et al., J. Mater. J. Mater. Cycles Waste 26 (2024) 1749–1760.
[158] L. Fang, F. Yan, J. Chen, X. Shen, Z. Zhang, ACS Sustain. Chem. Eng. 8 (2020) 6611–6621.
[159] B. Galey, B. Kim, D. Blanc, et al., Sustainable Chem. Pharm. 27 (2022) 100652.
[160] J. Wang, B. Zhang, Q. Xu, et al., J. Anal. Appl. Pyrolysis 174 (2023) 106132.
[161] X. Liu, Q. Tan, P. Wang, et al., Waste Manage. 184 (2024) 63–71.
[162] H. Wang, Z. Yang, X. Li, Y. Liu, Environ. Sci. Pollut. Res. 27 (2020) 17109–17122.
[163] G. Becker, D. Wust, H. Kohler, A. Lautenbach, A. Kruse, J. Environ. Manage. 238 (2019) 119–125.
[164] U. Roy, T. Radu, J. Wagner, J. Environ. Chem. Eng. 10 (2022) 107027.
[165] A. Hämäläinen, M. Kokko, V. Kinnunen, T. Hilli, J. Rintala, Water Res. 201 (2021) 117284.
[166] Q. Wang, C. Zhang, D. Patel, et al., Environ. Sci. Technol. 54 (2020) 8362–8372.
[167] L. Kong, X. Liu, Front. Environ. Sci. Eng. 14 (2020) 90.
[168] L. Paltrinieri, E. Huerta, T. Puts, et al., Environ. Sci. Technol. 53 (2019) 2396–2404.
[169] Y. Bian, X. Chen, L. Lu, P. Liang, Z.J. Ren, ACS Sustain. Chem. Eng. 7 (2019) 7844–7850.
[170] M. Kobya, P. Omwene, S. Sarabi, S. Yildirim, Z. Ukundimana, Process Saf. Environ. Prot. 152 (2021) 188–200.
[171] J. Ano, A. Assemian, Y. Yobouet, et al., Process Saf. Environ. Prot. 129 (2019) 184–195.
[172] Y. Ren, W. Zheng, X. Duan, et al., Environ. Funct. Mater. 246 (2022) 120699.
[173] B.I. Ojoawo, D.A. Daramola, Resour. Conserv. Recycl. Adv. 20 (2023) 200194.
[174] J. Wang, M. Li, A. Guan, et al., J. Hazard. Mater. 429 (2022) 128001.
[175] Y. Takabe, N. Ota, M. Fujiyama, et al., Sci. Total Environ. 706 (2020) 136090.
[176] N. Semerci, B. Kunt, B. Calli, Int. Biodeterior. Biodegrad. 144 (2019) 104739.
[177] S. Shaddel, S. Ucar, J. Andreassen, S. Osterhus, Water Sci. Technol. 79 (2019) 1777–1789.
[178] E.I. Epelle, A. Macfarlane, M. Cusack, et al., J. Microbiol. Methods 194 (2022) 106431.
[179] B. Yang, X. Li, Z. Lin, et al., Waste Manage. 102 (2020) 751–762.
[180] S. Chen, J. Gao, B. Dong, Sci. Total Environ. 759 (2021) 143573.