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Current status and perspectives on anaerobic co-digestion and associated downstream processes

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Current status and perspectives on anaerobic co-digestion and associated downstream processes

Abstract

Anaerobic co-digestion (AcoD) has the potential to utilise spare digestion capacity at existing wastewater treatment plants to simultaneously enhance biogas production by digesting organic rich industrial waste and achieve sustainable organic waste management. While the benefits of AcoD regarding biogas production and waste management are well established, the introduction of a new organic waste (i.e. co-substrate) with different chemical composition compared to residential sewage sludge is expected to impact on not only the anaerobic digestion process itself but also downstream processing of biogas and digestate. This work critically evaluates the potential impact (both positive and negative) of co-digestion on key downstream processes in the context of AcoD of sewage sludge and organic waste. AcoD can potentially lead to significant changes in biogas quality, digestate dewaterability, biosolids odour and the nutrient balance within the overall wastewater treatment process. The literature reviewed here suggests that effective management of these impacts can enhance the economic and environmental benefits of AcoD. Potential techniques to manage the impact of AcoD on downstream processing include co-substrate selection to minimise sulphur content, co-substrate pretreatment to improve dewaterability, process optimisation to minimize downstream impacts, biological desulphurisation of biogas, and side stream nutrient recovery. These techniques have been investigated and in some cases successfully applied for conventional anaerobic digestion. Nevertheless, further research is needed to adapt them for AcoD. In particular, the issue of nutrient accumulation due to AcoD can be seen as an opportunity to utilise recently commercialised technologies (e.g. Phosnix and Ostara) and currently emerging processes (e.g. forward osmosis and electrodialysis) for phosphorus recovery from food waste and wastewater.

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Current status and perspectives on anaerobic co-digestion and associated downstream processes

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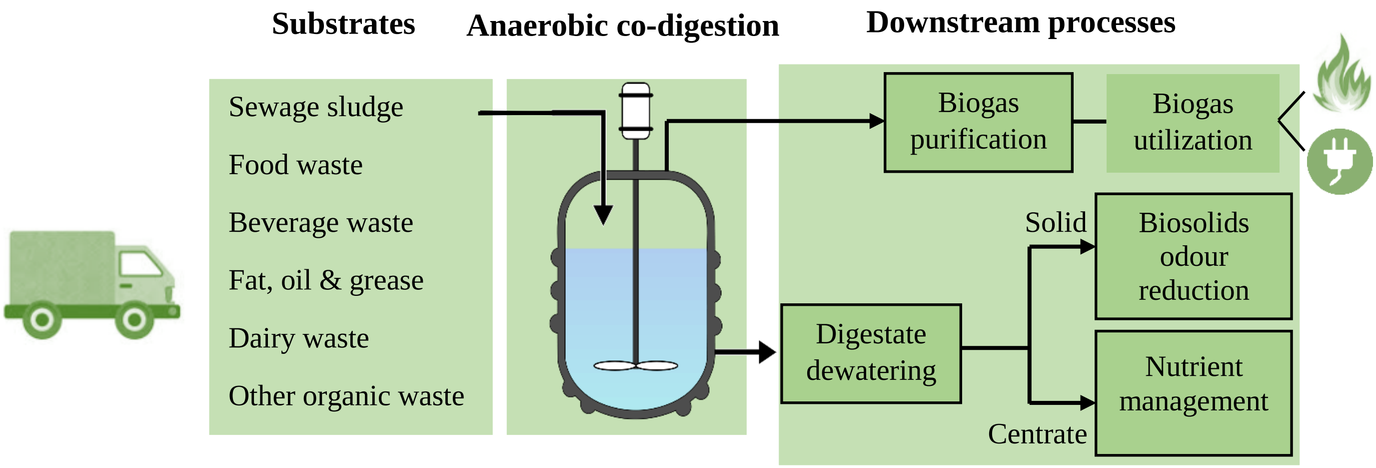
Abstract

Anaerobic co-digestion (AcoD) has the potential to utilise spare digestion capacity at existing wastewater treatment plants to simultaneously enhance biogas production by digesting organic rich industrial waste and achieve sustainable organic waste management. While the benefits of AcoD regarding biogas production and waste management are well established, the introduction of a new organic waste (i.e. co-substrate) with different chemical composition compared to residential sewage sludge is expected to impact on not only the anaerobic digestion process itself but also downstream processing of biogas and digestate. This work critically evaluates the potential impact (both positive and negative) of co-digestion on key downstream processes in the context of AcoD of sewage sludge and organic waste. AcoD can potentially lead to significant changes in biogas quality, digestate dewaterability, biosolids odour and the nutrient balance within the overall wastewater treatment process. The literature reviewed here suggests that effective management of these impacts can enhance the economic and environmental benefits of AcoD. Potential techniques to manage the impact of AcoD on downstream processing include co-substrate selection to minimise sulphur content, co-substrate pretreatment to improve dewaterability, process optimisation to minimize downstream impacts, biological desulphurisation of biogas, and side stream nutrient recovery. These techniques have been investigated and in some cases successfully applied for conventional anaerobic digestion. Nevertheless, further research is needed to adapt them for AcoD. In particular, the issue of nutrient accumulation due to AcoD can be seen as an opportunity to utilise recently commercialised technologies (e.g. Phosnix and Ostara) and currently emerging processes (e.g. forward osmosis) for phosphorus recovery from food waste and wastewater.

Keywords: Anaerobic co-digestion; Biosolids quality; Downstream processes; Organic wastes; Process stability; Nutrient recovery.

Graphical abstract



1. Introduction

Energy security, resource depletion and environmental protection are significant challenges of our time [1]. Efforts to address these challenges has resulted in a paradigm shift in organic waste management towards the circular economy concept [2]. A fundamental aspect in this paradigm shift is the diversion of organic waste from landfilling, which is a prevalent method for waste disposal in the developing world and even some of developed countries including Australia and the United States [3]. The real cost of solid waste disposal by landfilling is high when taking into account the land value, the cost of transportation, landfilling operation, and leachate treatment, post-closure maintenance, environmental pollution, and resource depletion. In this context, anaerobic co-digestion (AcoD) is a pragmatic platform for simultaneous resource recovery and sustainable management of organic wastes.

In the urban environment, AcoD can be described as the utilisation of spare digestion capacity in existing wastewater treatment plants (WWTPs) to co-digest organic wastes and sewage sludge [4]. There are two main drivers for the development and uptake of co-digestion by the water industry, namely reducing electricity costs by producing renewable electricity and also generating extra revenue by processing industrial organic waste. AcoD can help achieve simultaneous energy recovery and sustainable waste management for the water industry. Anaerobic digesters at most WWTPs are operated at a low organic load rate, often well below 1 kg volatile solids (VS)/m^{3}d [5]. Thus, by utilising co-digestion in WWTPs, revenue from gate fees or service charges, energy production, and reduction in greenhouse gas emission can be realised with minimal capital investment.

AcoD also offers a potential platform for the recovery of augmented nutrients (phosphorus and nitrogen) derived from food waste and other nutrient-rich co-substrates. In other words, following AcoD, integrated technologies can be applied to extract the liberated carbon, nitrogen and phosphorus to realise efficient energy and resources recovery [6, 7]. It is noteworthy that among all available options for food waste management, AcoD has been reported to have the least impact in terms of equivalent green-house gas emission [8].

To date, there have been very few comprehensive experimental assessments of the impacts of AcoD on downstream processing of anaerobic digestion products. In this review, the effects of AcoD on biogas quality and upgrading, digestate dewatering properties, biosolids odorous emissions and nutrient management will be systematically discussed. Techniques to manage the effects on downstream processing are also reviewed in the context of conventional anaerobic digestion and the potential to adapt them for AcoD applications.

2. Downstream processing of AD products

Biogas and digestate are primary products from anaerobic digestion. In most cases, further downstream processing is required for their beneficial use or safe disposal. Key downstream processes to purify and utilise biogas and to manage digestate are summarised in Figure 1. The operation and maintenance of these downstream processes account for a large portion of the ongoing cost of the overall AD process. Therefore, any changes to these downstream processes may alter the overall economic outlook of AcoD. Biogas contains mostly methane and, thus, it is a valuable biofuel. Biogas also contains several other gases such as water vapor, carbon dioxide, hydrogen sulfide, and ammonia that either do not have any energy value or can interfere with biogas utilization. Digestate contains 2 to 6% of solids with the balance being water. As a result, solid/liquid separation (also known as dewatering) is an essential step to reduce the cost of transportation and facilitate cost-effective digestate management.

[FIGURE 1]

Co-substrate addition can positively or negatively impact not only the AcoD process but also downstream processing of biogas and digestate (Figure 2). For example, AcoD digestion of sewage sludge and a carbon rich co-substrate (such as soft drink waste) may lead to a lower

H_2S content in biogas due to the dilution effect. On the other hand, an increase in H_2S content in biogas is expected when co-digesting sewage sludge and high protein food waste [9]. Given the significant operational cost of downstream processes associated with AD, enhancing positive impacts and mitigating negative ones are essential to the economics and environmental sustainability of AcoD.

[FIGURE 2]

2.1. Biogas quality

Biogas purification is necessary to remove unwanted gases (e.g. H_2S, water vapour, ammonia) and increase its thermal value. In the combustion process, H_2S is converted into SO_2, which is a highly corrosive gas to plant equipment. Thus, for electricity production, hydrogen sulfide removal is necessary to avoid long term corrosion to power generation equipment. In addition, the removal of other gases including carbon dioxide, ammonia and siloxanes is required for upgrading to biomethane. Upgrading to biomethane provides significantly more added-value to the produced biogas since biomethane can be used as transport fuel, town gas, and even used as raw materials for the chemical industry. Biogas purification into biomethane is complex and, thus, is only economically viable at large scale and with a strong demand for biomethane. As a result, full scale biogas purification into biomethane is limited and is mostly restricted to Europe where energy policies are favourable [10].

Toxic effects of inhibitory compounds from co-substrates can lead to AcoD process instability, reflected by accumulation of volatile fatty acids (VFAs), reduction in VS conversion to biogas, and hence, biogas production. AcoD process instability can also directly impact biogas quality (i.e. low CH_4 and high H_2S contents). For example, phenolic compounds which occur naturally in some organic products such as olive oil and red wine can inhibit the anaerobic process. Up to 50% reduction in biogas production and a decrease in CH_4 content have been observed and attributed to phenolic inhibition at phenolic concentration ranging from 120 to 594 mg·L^{-1}, depending on autoxidation, polarity, type, size and number of phenolic compounds [11].

2.2. Digestate dewaterability

Digestate dewatering involves pre-conditioning by polymer addition followed by a physical separation technique. Thus, both polymer demand and digestate dewaterability are important. Polymer (e.g. polyacrylamide) addition in the pre-conditioning step improves digestate dewaterability by reducing the specific resistance to filtration. Several techniques including screw press, centrifuge, and belt press can then be used for dewatering. A typical dewatering process can achieve the final biosolids with 15 to 30% solid content [8, 12].

Performance of the dewatering process are governed by several inter-related factors, most notably the organic and inorganic content of the digestate (Table 1). In general, polymer demand increases and dewaterability decreases as the content of extracellular polymeric substances (EPS) of VS in the digestate increases. EPS has a high affinity to water; thus, water can be captured inside EPS-rich flocs [13]. EPS content also controls the physico-chemical and biological properties of flocs (e.g. surface charge, rheological behavior, and disintegration of flocs) [13], which are directly related to digestate dewaterability.

When the microbial system is under stress during anaerobic digestion, more EPS is generated. The stress condition also leads to a low VS removal efficiency, thus, a high VS content in the digestate. Skinner et al. [14] have observed a positive correlation between digestate dewaterability and volatile solid content. Similarly, Girault et al. [15] reported that the volatile solid/total solid ratio governs the dewatering efficiency of digested sludge.

Digestate dewatering is also governed by chemical composition and physical configuration of the aggregates or flocs. Since the primary purpose of polymer addition is to reduce the surface charge of digestate particles to facilitate inter-particle (including EPS) bridging and improve dewaterability [16], physicochemical parameters including the presence of multivalent cations, surface charge of the sludge particles, pH and salinity can also influence the dewatering process (Table 1).

[TABLE 1]

Organic loading applied in co-digestion is higher than that in mono-digestion [4]. On the other hand, organic over loading during AcoD can cause accumulation of intermediate inhibitory compounds such as VFAs and long chain fatty acids (LCFAs). In turn, the accumulation of VFAs and LFCAs in the digestate can increase polymer demand and reduce digestate dewaterability due to a high content of EPS and VS. In addition, the addition of undegraded lipids into the digestate sludge due to high co-substrates ratio can also lead to an increase in polymer demand during the preconditioning stage [9], thus, negatively affecting digestate dewatering [15]. It is noteworthy that Higgins et al., [9] reported an increase in polymer demand due to food waste co-digestion, however, the increase in polymer demand also resulted in a substantial increase in the final solid cake content (in other words, an improved digestate dewaterability).

2.3. Nutrients in sludge centrate

The liquid stream (often called filtrate or sludge centrate) from digestate dewatering is rich in both phosphorus and ammonium nitrogen [3, 17]. In a conventional WWTP, the sludge centrate is returned to the plant inlet. This practice could lead to nutrient accumulation and potentially struvite blockage and high nutrient content in the effluent. In recent years, there has been a growing interest to recover these nutrients from the sludge centrate not only to reduce struvite blockage and the treatment demand for phosphorus removal but also to provide a sustainable source of fertilizers for agricultural production.

Interest in nutrient recovery from wastewater has spurred successful commercialisation of several trademark processes. Notable examples are Phosnix and Ostara Pearl [18]. Phosnix is a side stream process that enables phosphorus removal and recovery from the sludge centrate as granulated struvite. The sludge centrate is fed into the bottom of a fluidized bed reactor, containing a bed of granulated struvite as a seed material for crystal growth. By adding magnesium hydroxide to achieve magnesium to phosphate ratio of 1:1, adjusting pH to 8.2-8.8 and the crystal retention time to 10 days, pellets between 0.5 and 1.0 mm in diameter can be harvested from the bottom of the reactor column. Finer granules of struvite in the separated liquid are returned to the column to provide additional seed material to assure process continuity. Ostara Pearl is a chemical crystallization process in an up-flow fluidized bed reactor with multiple reactive zones of increasing diameters. This process allows for the production of large struvite pellets with 1.5 to 4.5 mm in diameter, while maintaining fine crystal nuclei from the top of the reactor. Struvite crystallization is controlled by a combination of magnesium dose, pH control and by means of a treated effluent recycle. Crystal Green (containing 5% nitrogen, 28% phosphorus, and 10% magnesium) is a trade mark product of the Ostara process and can be used as slow release fertilizer.

Magnesium addition is required for struvite recovery from sludge centrate. Given the market price of $MgCl_2$ of over 100 USD/ton [19], the current market value of struvite as phosphorus fertilizer is not sufficient to recover operating cost. Indeed, prevention of phosphorus build-up, which can subsequently result in excessive struvite blockage and high phosphorus content in the effluent, is still the key justification for phosphorus recovery facilities at WWTPs. There has been some recent effort to improve the economics of phosphorus recovery by enriching the phosphorus content in sludge centrate prior to chemical precipitation via forward osmosis [7]. The forward osmosis process can provide phosphorus rich solution, thus optimizing the stoichiometric ratio for crystallization, improving precipitation kinetics and lowers the chemical (i.e. magnesium salts and caustic) demand.

The sludge centrate also has a significant ammonium nitrogen content ranging from 741 to 4000 mg·L^{-1} as ammonia [20-22]. Ammonia is a major pollutant although its recovery is not as important as phosphorus since nitrogen fertilizer can be readily produced by the Haber-Bosch process [19, 23]. In a typical WWTP, the sludge centrate is returned to the head of work and ammonium nitrogen is then removed via biological nitrification and denitrification. Here, opportunities exist for a side stream treatment process such as air stripping [24, 25] and anammox [26] to specifically target nitrogen rich sludge centrate. In particular, anammox has emerged as an energy efficient process, where nitrite and ammonium are directly converted to nitrogen [27, 28]. A number of full scale anammox plants for nitrogen removal have been recently reported by Speth et al., [29].

AcoD presents both new challenges and opportunities to nutrient management. Sludge centrate from the dewatering of digestate originated from co-digestion with protein rich co-substrate (e.g. food waste) typically contains higher nitrogen and phosphorus concentrations compared to mono-digestion. If these nitrogen and phosphorus are returned to the inlet of the WWTP, there is a risk of nutrient overloading which can disrupt plant operation. Nutrient build-up can also entail the need for additional treatment to meet the nitrogen and phosphorus standards in the effluent [5]. On the other hand, it is economically more favourable to recovery nitrogen and phosphorus from nutrient rich sludge centrate. Indeed, the recovery of phosphorus (and to a lesser extent nitrogen) is critical not only for AcoD operation but also for resource preservation since they are essential for agriculture production [30].

2.4. Biosolids quality

After dewatering, the obtained biosolids can be beneficially reused via land application or disposed by incineration. Land application is an environmentally friendly biosolids management approach and is widely practiced in Australia and the USA where agricultural land for non-edible crops is readily available. In Europe, Japan, and several other countries where the availability of space is limited, incineration has become an alternative for biosolids disposal [5]. Incineration is more favorable where concerns about biosolids odour and the occurrence of pathogenic agents in biosolids outweigh those about air pollution. Regardless of the final disposal options, the efficiency of the dewatering process is important for cost-effective management of the biosolids since a high solid content can reduce the cost of transportation for land application and consumption of auxiliary fuel for incineration.

When incineration can be used for digestate disposal, the solid cake content is the most important parameter. As a result, after dewatering, thermal drying is often used to further increase the solid cake content to as much as 70%. For land application, odorous emission is arguably the most important factor influencing beneficial reuse or disposal options of biosolids. Depending on its quality, biosolids can be used for land application for forestation, disturbed land in need of reclamation and even agricultural production. The impacting points of the released malodour during the typical operation are therefore restricted to transportation and land application activities themselves.

Biosolids odour can be a major roadblock for beneficial reuse of the solid residuals from anaerobic digestion. These odorants primarily include six types of volatile compounds including sulfur bearing compounds, nitrogen bearing compounds, VFAs, ketones, aldehydes, and hydrocarbons [31, 32]. Most studies to date have focused on sulfur bearing compounds when assessing biosolids odour emissions since they are the primary constituents of odour from biosolids following anaerobic digestion of protein-rich substrates [33, 34]. These studies have provided important insights to connect the issue of biosolids odour and AcoD with sulfur rich co-substrates [35]. Of a particular note, Higgins et al., [36] proposed a cyclic pathway to describe the production and transformation of volatile sulfur compounds and H_2S. They hypothesize that the addition of sulfur rich co-substrates during AcoD can result in an increase in biosolids odorous emission. This involves processes such as degradation of protein, generation of associated volatile organic sulfur compounds (e.g. methanethiol) and subsequent formation of H_2S [4, 36, 37].

The sensorially relevant volatile compounds are predominantly generated from microbial degradation of organic matter [38, 39]. p-cresol, trimethylamine and VFAs have been demonstrated as key sensorially relevant volatile compounds that are associated with anaerobic degradation of organic matter [31, 39]. In addition, a range of aromatic compounds (e.g. toluene, p-cresol and indole) can be generated through anaerobic degradation [32, 40]. Although detected at low levels (i.e. <500 $\mu g/m^3$), these aromatic compounds are considered to be important odorants when peaks of volatile sulfur compounds diminish [40].

Biosolids odour is also used as a surrogate to indicate microorganism regrowth or the occurrence of pathogenic microorganisms in biosolids. In other words, the release of biosolids odour can be an indicator for pathogenic regrowth [31]. Key factors governing occurrence of pathogenic microorganisms in biosolids include digestion temperature and substrate availability [41-43]. Thermophilic digestion is thought to have higher pathogen inactivation compared with mesophilic digestion. However, it has been established that microbial competition for substrates rather than temperature is responsible for pathogen reduction in biosolids during anaerobic digestion [44]. Substrate availability is also a major factor influencing the regrowth of pathogens after digestate dewatering [45].

3. Addressing the impact of AcoD on downstream processes

Downstream processes are interrelated. Thus, any changes to one process can influence another downstream process in the overall system (Figure 3). For instance, pre-treatment of substrates can enhance VS destruction, which can also improve digestate dewaterability. The improved dewaterability can in turn mitigate odorous emission through a reduction in the occurrence of volatile organic compounds in biosolids.

[FIGURE 3]

3.1. Biogas quality, digestate dewaterability and biosolids odorous emissions

In general, positive correlations are expected between improved biogas quality, enhanced digestate dewaterability and mitigated biosolids odorous emissions. Key strategies to manage the impact of AcoD on downstream processes are summarized in Figure 3 and discussed below.

3.1.1. Co-substrate selection

The variations in co-substrate properties and composition govern the AcoD process (particularly biogas quality), and thus the downstream processes. In some cases, co-digestion can lead to synergistic effects, reflected by either a boost in specific methane yield or an increase in biogas production kinetics [4, 46]. The synergistic effects are often associated with higher volatile solid removals and improved biogas quality (e.g. higher methane content), likely associated with the balanced C/N ratio and dilution of inhibitory and toxic substances [47, 48]. The extent of the synergistic effects can be notable during AcoD [47-51]. In contrast, antagonistic effects have also been observed during AcoD of sewage sludge and organic waste [52]. Antagonistic effects have negative impact on overall downstream processes. It is noteworthy that the types of carbon source from metabolic degradation of co-substrates during AcoD may also affect methanogenic activities and pathways, thus leading to varied biogas quality [53].

Co-substrates can be broadly classified into carbohydrate-rich, protein-rich and lipid-rich organic materials. At high organic loading, carbohydrate-rich co-substrates can be easily degraded, causing the accumulation of VFAs, and induce process instability. Protein-rich co-substrates can be degraded into two most predominant forms of inorganic nitrogen: ammonium-N and free ammonia during AcoD. Depending on the operational factors (i.e. pH and temperature), high concentrations of free ammonia can inhibit methanogens, leading to process instability, system failure and deteriorated downstream processes [4, 54]. Lipid-rich co-substrates have a high biogas yield, however, an elevated level of intermediate products (i.e. long chain fatty acids) can suppress methanogenic microbial population, and lead to VFA accumulation, system instability and negative downstream impact. Thus, it is important to avoid co-substrate organic overloading during AcoD [52].

Coagulants (e.g. aluminum or ferric salts) addition to sewage sludge can reduce the generation of total volatile organic sulphur compounds [55, 56]. Indeed, these coagulants can react with sulphide to form precipitate. Thus, metal addition (e.g. iron and aluminum) can be used for hydrogen sulfide control, and subsequently total volatile organic sulphur compounds control, as total volatile organic sulphur compounds can be generated by microbial conversion of hydrogen sulfide to methanethiol [36, 57]. It is noteworthy that by supplementing iron (e.g. Fe^0, Fe(II) and Fe(III)) to sewage sludge during AcoD, the release of bound proteins for further biodegradation can be expected due to iron’s action as electron donor/acceptor and micronutrient (i.e. cofactor of key enzymatic activities) [58, 59]. Thus, there is possible increase in volatile solid destruction and process performance at the presence of iron addition during AcoD. Indeed, Vrieze et al. [60] observed more stable methane production during AcoD of kitchen waste and sludge amended with Fe(II), compared to mono-digestion of kitchen waste.

3.1.2. Co-substrate pre-treatment

Digestate dewaterability can be improved by converting bound water in the solid particles into free water. The reduction in bound water in digestate can reduce viscosity and thus dewaterability. Pre-treatment methods, such as thermal treatment [61-63], chemical treatment [64, 65], and biological treatment [66], have been employed to facilitate digestate/sludge dewatering.

Thermal pretreatment can increase the final solid content of biosolids, possibly due to the reduction in the bound water in digestate and thus viscosity of the digestate [67]. At a higher thermal pretreatment temperature, a greater solubilization of organics occurs. This allows the remaining materials to be smaller in size and less amenable to removal by polymer conditioning, as polymer conditioning can mainly remove biocolloids in the range of up to 1.5 µm [61, 68]. Studies have been performed to evaluate the effect of thermal hydrolysis pretreatment temperature on subsequent digestion performance and operation, as well as downstream parameters such as dewatering and cake quality [61, 63]. The authors found that higher thermal hydrolysis temperatures improved the volatile solid reduction and biogas quality, as well as cake solids after dewatering [61]. In addition, thermally pretreated co-substrates (e.g. microalgae) can facilitate the release of nutrients during AcoD with sewage sludge [69]. Other pretreatment methods such as thermo-oxidative technique can also enhance the removal of volatile sulfur compounds and lead to optimised AcoD processes and mitigation in biosolids odorous emissions [70].

Chemical pretreatment can have additional effects on facilitating sludge dewatering and biogas production when combined with thermal pretreatment [71]. Multivalent cations (such as Ca^{2+}) exhibit positive effects on improving sludge dewaterability by forming hydroxybases or changing digestate/sludge properties as surface charge, viscosity and floc strength [72, 73]. Recently, Song et al. [65] applied a combination of persulfate and zero valent iron to enhance the dewaterability of anaerobically digested sludge. Chemical pretreatments can also be used to facilitate downstream nutrient recovery. For example, Shi et al. [74] used acidified pig manure to recover nitrogen, phosphorus and VFAs using bipolar membrane electrodialysis[74].

Biological pretreatment is a promising technique to improve biosolids dewaterability, although this has not been applied at full scale. Murugesan et al. [75] used an *Acidithiobacillus ferrooxidans* culture to pre-condition the digestate and observed a noticebale decrease in extractable EPS content. They suggested a possible application via bioacidification treatment to improve sludge dewaterability [75].

3.1.3. Process optimisation

Operating parameters including temperature, retention time and organic loading can be optimized to tackle the negative downstream processes. For example, dewaterability of mesophilic digestate is usually better than that of thermophilic digestate due to better process stability at mesophilic conditions [76]. On the other hand, an optimised AcoD process with adequate solid retention time and organic loading can enhance volatile solid destruction. This leads to an enhanced VFA yield and specific methane yield, thus reducing biosolids odorous emissions. As VFA concentrations can be empirically corrected with the odor production potential [38], not only is the enhanced VFA production during AcoD process beneficial for subsequent VFAs recovery or methanogenesis processes, but it also is pivotal in reducing biosolids odorous emissions [77]. In some instances, it can be achieved by coupling the AcoD process with a bioelectrochemical systems (e.g. microbial electrolysis cell) to simultaneously recover VFAs and ammonia, as well as avoid AcoD system instability [78]. In addition, optimised process design can improve digestate dewaterability. For example, Cobbledick et al. [79] explored the possibility of applying recuperative thickening technology for enhancing biogas production and dewaterability in anaerobic digestion processes. It is noteworthy that recuperative thickening technology can be employed in conjunction with AcoD to enhance volatile solid destruction and the methane yield [79].

3.1.4. Biological desulphurization

Biological desulphurization process can be used to limit the formation of hydrogen sulfide in biogas. Biological desulphurization can be achieved by regulating the redox potential in the digester using a small amount of an oxidizer (such as oxygen or nitrate) to prevent the reduction of sulphur to hydrogen sulfide or to oxidise hydrogen sulfite to elementary sulphur by biocatalysts from specific S oxidizing microorganisms such as *Thiomicrospira sp.* or *Thiobacillus sp* [80]. Nghiem et al., [81] have successfully applied this approach through an engineered technique to regulate the oxidation reduction potential in anaerobic digester to inject a minute amount of oxygen into a pilot anaerobic digester. They demonstrated a complete inhibition of hydrogen sulfite formation (thus no hydrogen sulfide in biogas), while no discernible changes in biogas production or composition can be observed [81]. There are several other options, including the use of a commercial rubber waste product or fine rubber particle media can be adopted [82]. In this approach, hydrogen sulfide is possibly removed via adsorption to carbon black (a carbonaceous material used in rubber products) and reaction with zinc oxide to form stable zinc sulfide. Both carbon black and zinc oxide are abundant in commercial rubber waste [82].

3.2. Nutrient management

There have been several excellent reviews on nutrient (particularly phosphorus) recovery from digestate [7, 30, 83-85]. Technologies for phosphorus removal and recovery can be broadly divided into four major categories, including chemical precipitation, enhanced biological treatment, crystallisation, and membrane based technologies (Table 2).

Chemical precipitation is arguably the most versatile phosphorus removal technique (Table 2). Ferric chloride ($FeCl_3$) and ferrous sulphate ($FeSO_4$) are the most used iron salts to remove phosphorus, with the former salt being the preferred one due to the oxidation of Fe^{2+}. The application of $FeCl_3$ during the sludge dewatering process can inhibit struvite formation in the anaerobic sludge digester and sludge lines due to the formation of ferric phosphate which can be captured in the sludge cake due to its low solubility [86]. In addition, Fe:P molar ratio of more than 1 is required due to the formation of the by-product iron hydroxides. Reported optimal pH conditions fluctuate due to the variation in physicochemical characteristics of digestate liquor and different procedures and conditions used for phosphorus precipitation.

Phosphorus can also be removed and recovered biologically via its incorporation in the biomass. Microorganisms capable of accumulating polyphosphate (Poly-P) can take up excess phosphorus under alternating anaerobic and aerobic/anoxic conditions, thus, allowing for phosphorus accumulation in biomass. This process is often referred to as enhanced biological phosphorus removal (EBPR). In this process, Poly-P accumulating organisms (PAOs) can recovery dilute dissolved P in addition to other cations (e.g. Mg^{2+} and K^+) in the liquid form and concentrate them in the sludge biomass in the form of intracellular Poly-P [30]. The recovery of phosphorus is completed by the chemically precipitating dewatered P rich sludge as fertilizer. EBPR process often requires external carbon source for microbial metabolism. Thus digestate from anaerobic acidification of sewage sludge and organic wastes can be an excellent supply of carbon source to the EBPR process [87, 88]. The EBPR process is often coupled with ammonia oxidation due to the cohabitation of ammonia oxidizing bacteria and PAOs in wastewater treatment systems [89].

Crystallisation technologies have been applied at several full scale plants to recover P in the form of struvite ($MgNH_4PO_4{\cdot}6H_2O$), and to a lesser extent, hydroxyapatite ($Ca_{10}(PO_4)_6(OH)_2$) or calcium phosphate ($Ca_3(PO_4)_2$) [90-92]. The key driver for phosphorus recovery in these plants is to prevent uncontrolled struvite scaling or struvite blockage within the plant [90, 91]. It is noteworthy that, at the presence, revenue from struvite recovery (given the current market value of phosphorus fertilizer) is not sufficient to recover the cost of this process. Thus, further optimisation is necessary to improve the efficiency and economic viability of phosphorus recovery if crystallization technologies are adapted for AcoD applications. The co-precipitation of calcium, ferric, aluminium [93, 94] and organic compounds [21, 95, 96] significantly reduces the recovery and purity of struvite and inhibits the crystal growth due to the confined space where crystals could be formed [97, 98]. Hence, liquid matrix can undergo some pre-treatment (e.g. acid leaching, chelating agents and microwave treatment) to minimise the inhibitory effect prior to phosphorus recovery via struvite precipitation [99, 100].

The efficiency of crystallisation for phosphorus recovery is directly proportional to the concent of phosphorus in the sludge centrate. For example, due to competition from impurities in the centrate, the chemical cost for phosphorus recovery increases exponentially as the phosphorus concentration in the initial feedstream decreases. Thus, innovative solutions, such as membrane based technologies, are needed to improve the economic feasibility of struvite precipitation process (Table 2). Membrane based technologies, in particular forward osmosis and electrodialysis, have the potential to simultaneously produce high quality effluent and pre-concentrated wastewater to facilitate nutrient recovery [7, 101]. For example, Xie et al. [20, 102] developed a novel forward osmosis - membrane distillation hybrid system for extracting phosphorus from sludge centrate in the form of struvite. Forward osmosis can concentrate orthophosphate and ammonium for subsequent phosphorus recovery with higher crystal growth kinetics. On the other hand, Shi et al. [74] used bipolar membrane electrodialysis to recover ammonium, phosphate and VFAs simultaneously from both synthetic and real pig manure hydrolysate.

[TABLE 2]

4. Conclusion

Data corroborated in this review demonstrates an array of potential impacts of AcoD on downstream processing including deteriorated biogas quality, variation in digestate dewaterability, biosolids odour, and nutrient buildup. This work also reviews techniques that have been studied and in some cases successfully applied to conventional anaerobic digestion of a single substrate (such as sewage sludge) to improve the performance of downsteam processing. Strategies to adapt them for AcoD applications are then discussed. They include co-substrate selection, co-substrate pretreatment, process optimization, desulphurization, and nutrient recovery from the sludge centrate. In addition, integration with side-stream processes (e.g. forward osmosis) can potentially improve the economic of these processes for nutrient recovery. It is also highlighted in this review that downstream processes are interalated, thus, it is necessary to apply these techniques together within a holistic framework.

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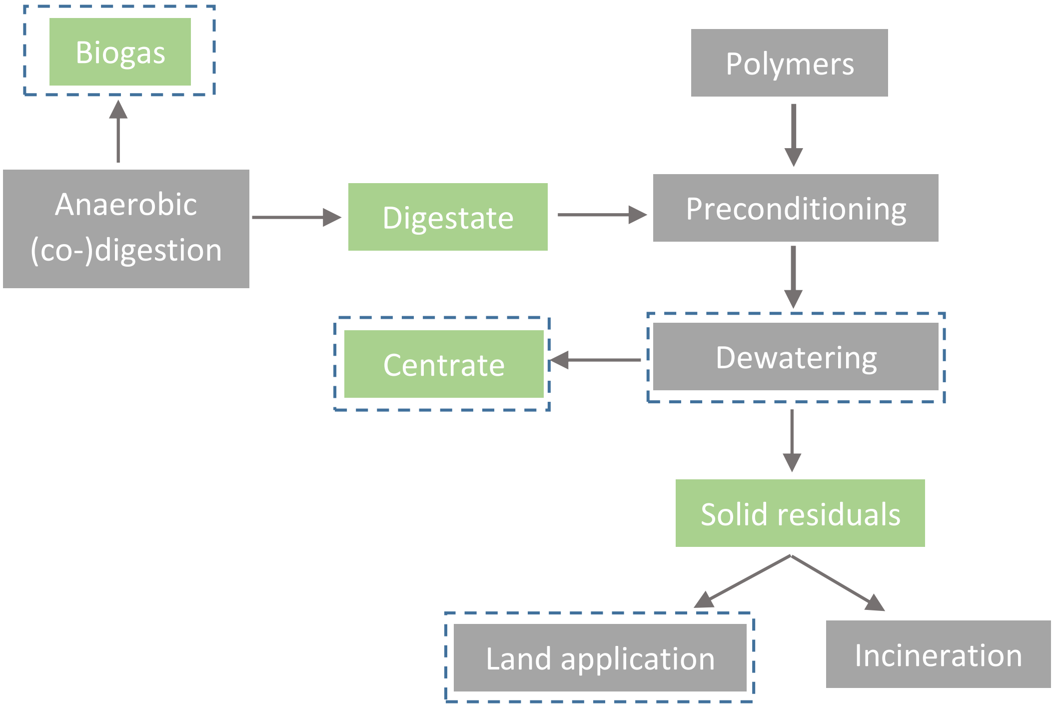

Figure 1: A schematic diagram for anaerobic digestion associated downstream processes shown in dashed squares: (1) digestate dewaterability; (2) biosolids odorous emissions; (3) nutrient management; and (4) biogas quality. Green-filled square denotes the byproducts from both anaerobic digestion and its downstream processes for bioenergy/resource recovery.

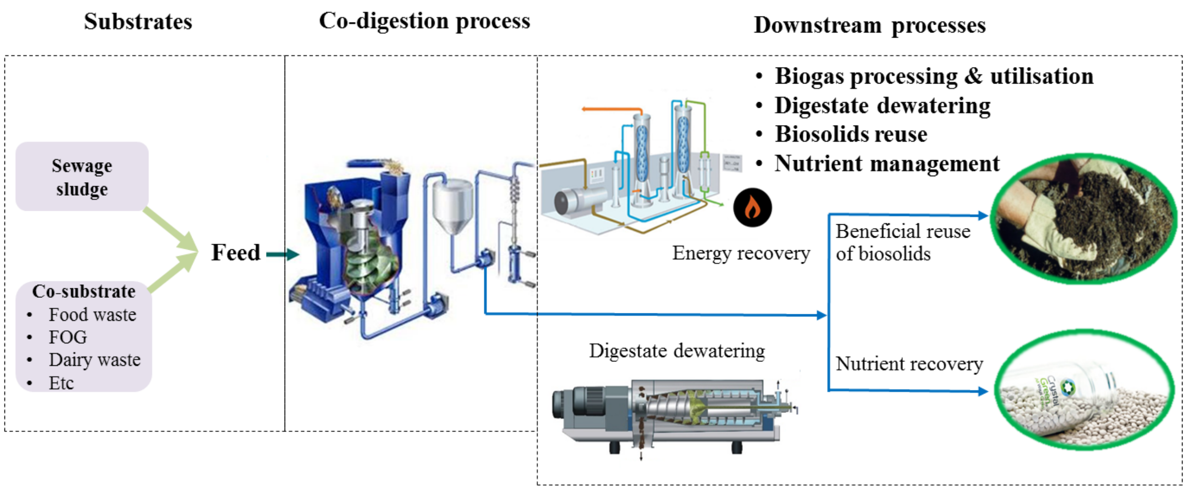

Figure 2: Possible impact of AcoD on downstream processing of biogas and digestate.

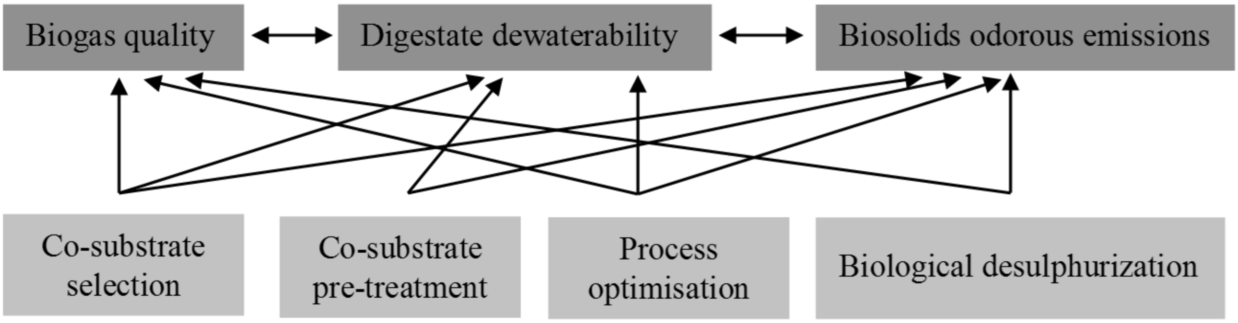

Figure 3: Perspectives on strategies to improve downstream processes: biogas quality, digestate, dewaterability, and biosolids odorous emissions. Correlated processes are denoted as ⟷.

Table 1: Factors negatively influencing digestate dewaterability.

Factors	**Effects**	**Mechanisms**
High EPS contents [13]	Increase polymer demand Reduce dewaterability	Water molecules can be captured inside EPS rich flocs, leading to an increase in polymer demand and poor digestate dewaterability
High volatile solid content [14]	Increase polymer demand Reduce dewaterability	EPS and volatile solid content are positively correlated
High content of multivalent cations [103]	Increase polymer demand	Charge neutralization is less effective
Highly charged sludge particles [104]	Increase polymer demand	More polymer is required for charge neutralization
Too acidic or basic [104]	Reduce dewaterability	Acidic/basic condition can alter flocs cohesion, facilitate polymeric structure breakdown, leading to poor dewaterability
High salinity [104]	Increase polymer demand	Charge neutralization is less effective

Table 2: Comparisons of P recovery techniques.

Techniques	**Advantages**	**Disadvantages**	**Technological readiness**
Chemical precipitation [86]	Mature technology	High chemical consumption (e.g. coagulant)	Available at full scale operation
Enhanced biological P removal [105, 106]	Reduced sludge production; Eliminating chemicals use; Can be easily integrated with existing operation	An addition process (e.g. chemical precipitation) is required for P recovery	Available at full scale operation
Crystallization technologies [18, 107]	Produce high value commercial products (e.g. struvite)	High capital and operational cost	Limited full scale operation
Membrane based technologies [108]	Can enhance P recovery; Modular and small physical footprint	Issues associated with membrane operation (e.g. salinity build-up, membrane fouling) and system scale-up	Lab scale demonstration