

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

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17 **Abstract**

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

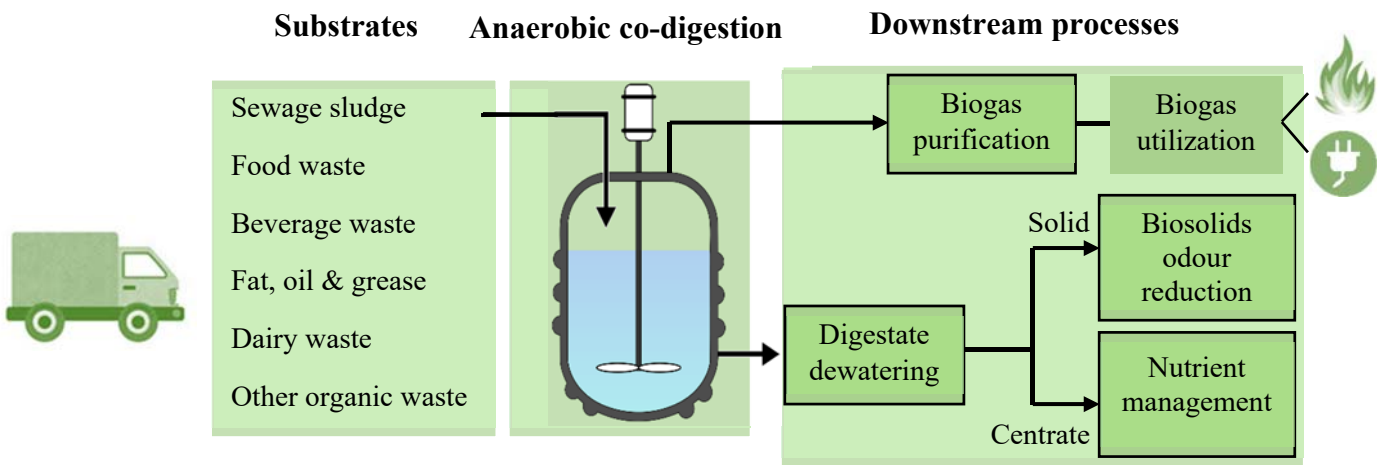
40 **Keywords:** Anaerobic co-digestion; Biosolids quality; Downstream processes; Organic
41 wastes; Process stability; Nutrient recovery.

42 **Graphical abstract**

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46 **1. Introduction**

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

57 In the urban environment, AcoD can be described as the utilisation of spare digestion
58 capacity in existing wastewater treatment plants (WWTPs) to co-digest organic wastes and
59 sewage sludge ⁴. There are two main drivers for the development and uptake of co-digestion
60 by the water industry, namely reducing electricity costs by producing renewable electricity
61 and also generating extra revenue by processing industrial organic waste. AcoD can help
62 achieve simultaneous energy recovery and sustainable waste management for the water
63 industry. Anaerobic digesters at most WWTPs are operated at a low organic load rate, often
64 well below 1 kg volatile solids (VS)/m³d ⁵. Thus, by utilising co-digestion in WWTPs,
65 revenue from gate fees or service charges, energy production, and reduction in greenhouse
66 gas emission can be realised with minimal capital investment.

67 AcoD also offers a potential platform for the recovery of augmented nutrients (phosphorus
68 and nitrogen) derived from food waste and other nutrient-rich co-substrates. In other words,
69 following AcoD, integrated technologies can be applied to extract the liberated carbon,
70 nitrogen and phosphorus to realise efficient energy and resources recovery ^{6, 7}. It is

71 noteworthy that among all available options for food waste management, AcoD has been
72 reported to have the least impact in terms of equivalent green-house gas emission ⁸.

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

79 **2. Downstream processing of AD products**

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

92 **[FIGURE 1]**

93 Co-substrate addition can positively or negatively impact not only the AcoD process but also
94 downstream processing of biogas and digestate (Figure 2). For example, AcoD digestion of

95 sewage sludge and a carbon rich co-substrate (such as soft drink waste) may lead to a lower
96 H₂S content in biogas due to the dilution effect. On the other hand, an increase in H₂S content
97 in biogas is expected when co-digesting sewage sludge and high protein food waste ⁹. Given
98 the significant operational cost of downstream processes associated with AD, enhancing
99 positive impacts and mitigating negative ones are essential to the economics and
100 environmental sustainability of AcoD.

101 [FIGURE 2]

102 2.1. Biogas quality

103 Biogas purification is necessary to remove unwanted gases (e.g. H₂S, water vapour, ammonia)
104 and increase its thermal value. In the combustion process, H₂S is converted into SO₂, which is
105 a highly corrosive gas to plant equipment. Thus, for electricity production, hydrogen sulfide
106 removal is necessary to avoid long term corrosion to power generation equipment. In addition,
107 the removal of other gases including carbon dioxide, ammonia and siloxanes is required for
108 upgrading to biomethane. Upgrading to biomethane provides significantly more added-value
109 to the produced biogas since biomethane can be used as transport fuel, town gas, and even
110 used as raw materials for the chemical industry. Biogas purification into biomethane is
111 complex and, thus, is only economically viable at large scale and with a strong demand for
112 biomethane. As a result, full scale biogas purification into biomethane is limited and is
113 mostly restricted to Europe where energy policies are favourable ¹⁰.

114 Toxic effects of inhibitory compounds from co-substrates can lead to AcoD process
115 instability, reflected by accumulation of volatile fatty acids (VFAs), reduction in VS
116 conversion to biogas, and hence, biogas production. AcoD process instability can also
117 directly impact biogas quality (i.e. low CH₄ and high H₂S contents). For example, phenolic
118 compounds which occur naturally in some organic products such as olive oil and red wine

119 can inhibit the anaerobic process. Up to 50% reduction in biogas production and a decrease in
120 CH₄ content have been observed and attributed to phenolic inhibition at phenolic
121 concentration ranging from 120 to 594 mg·L⁻¹, depending on autoxidation, polarity, type, size
122 and number of phenolic compounds ¹¹.

123 2.2. Digestate dewaterability

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

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

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

142 Digestate dewatering is also governed by chemical composition and physical configuration of
143 the aggregates or flocs. Since the primary purpose of polymer addition is to reduce the
144 surface charge of digestate particles to facilitate inter-particle (including EPS) bridging and
145 improve dewaterability¹⁶, physicochemical parameters including the presence of multivalent
146 cations, surface charge of the sludge particles, pH and salinity can also influence the
147 dewatering process (Table 1).

148 [TABLE 1]

149 Organic loading applied in co-digestion is higher than that in mono-digestion⁴. On the other
150 hand, organic over loading during AcoD can cause accumulation of intermediate inhibitory
151 compounds such as VFAs and long chain fatty acids (LCFAs). In turn, the accumulation of
152 VFAs and LCFAs in the digestate can increase polymer demand and reduce digestate
153 dewaterability due to a high content of EPS and VS. In addition, the addition of undegraded
154 lipids into the digestate sludge due to high co-substrates ratio can also lead to an increase in
155 polymer demand during the preconditioning stage⁹, thus, negatively affecting digestate
156 dewatering¹⁵. It is noteworthy that Higgins et al.,⁹ reported an increase in polymer demand
157 due to food waste co-digestion, however, the increase in polymer demand also resulted in a
158 substantial increase in the final solid cake content (in other words, an improved digestate
159 dewaterability).

160 2.3. Nutrients in sludge centrate

161 The liquid stream (often called filtrate or sludge centrate) from digestate dewatering is rich in
162 both phosphorus and ammonium nitrogen^{3,17}. In a conventional WWTP, the sludge centrate
163 is returned to the plant inlet. This practice could lead to nutrient accumulation and potentially
164 struvite blockage and high nutrient content in the effluent. In recent years, there has been a
165 growing interest to recover these nutrients from the sludge centrate not only to reduce struvite

166 blockage and the treatment demand for phosphorus removal but also to provide a sustainable
167 source of fertilizers for agricultural production.

168 Interest in nutrient recovery from wastewater has spurred successful commercialisation of
169 several trademark processes. Notable examples are Phosnix and Ostara Pearl ¹⁸. Phosnix is a
170 side stream process that enables phosphorus removal and recovery from the sludge centrate
171 as granulated struvite. The sludge centrate is fed into the bottom of a fluidized bed reactor,
172 containing a bed of granulated struvite as a seed material for crystal growth. By adding
173 magnesium hydroxide to achieve magnesium to phosphate ratio of 1:1, adjusting pH to 8.2-
174 8.8 and the crystal retention time to 10 days, pellets between 0.5 and 1.0 mm in diameter can
175 be harvested from the bottom of the reactor column. Finer granules of struvite in the
176 separated liquid are returned to the column to provide additional seed material to assure
177 process continuity. Ostara Pearl is a chemical crystallization process in an up-flow fluidized
178 bed reactor with multiple reactive zones of increasing diameters. This process allows for the
179 production of large struvite pellets with 1.5 to 4.5 mm in diameter, while maintaining fine
180 crystal nuclei from the top of the reactor. Struvite crystallization is controlled by a
181 combination of magnesium dose, pH control and by means of a treated effluent recycle.
182 Crystal Green (containing 5% nitrogen, 28% phosphorus, and 10% magnesium) is a trade
183 mark product of the Ostara process and can be used as slow release fertilizer.

184 Magnesium addition is required for struvite recovery from sludge centrate. Given the market
185 price of MgCl₂ of over 100 USD/ton ¹⁹, the current market value of struvite as phosphorus
186 fertilizer is not sufficient to recover operating cost. Indeed, prevention of phosphorus build-
187 up, which can subsequently result in excessive struvite blockage and high phosphorus content
188 in the effluent, is still the key justification for phosphorus recovery facilities at WWTPs.
189 There has been some recent effort to improve the economics of phosphorus recovery by
190 enriching the phosphorus content in sludge centrate prior to chemical precipitation via

191 forward osmosis ⁷. The forward osmosis process can provide phosphorus rich solution, thus
192 optimizing the stoichiometric ratio for crystallization, improving precipitation kinetics and
193 lowers the chemical (i.e. magnesium salts and caustic) demand.

194 The sludge centrate also has a significant ammonium nitrogen content ranging from 741 to
195 4000 mg·L⁻¹ as ammonia ²⁰⁻²². Ammonia is a major pollutant although its recovery is not as
196 important as phosphorus since nitrogen fertilizer can be readily produced by the Haber-Bosch
197 process ^{19, 23}. In a typical WWTP, the sludge centrate is returned to the head of work and
198 ammonium nitrogen is then removed via biological nitrification and denitrification. Here,
199 opportunities exist for a side stream treatment process such as air stripping ^{24, 25} and
200 anammox ²⁶ to specifically target nitrogen rich sludge centrate. In particular, anammox has
201 emerged as an energy efficient process, where nitrite and ammonium are directly converted to
202 nitrogen ^{27, 28}. A number of full scale anammox plants for nitrogen removal have been
203 recently reported by Speth et al., ²⁹.

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

214 2.4. Biosolids quality

215 After dewatering, the obtained biosolids can be beneficially reused via land application or
216 disposed by incineration. Land application is an environmentally friendly biosolids
217 management approach and is widely practiced in Australia and the USA where agricultural
218 land for non-edible crops is readily available. In Europe, Japan, and several other countries
219 where the availability of space is limited, incineration has become an alternative for
220 biosolids disposal⁵. Incineration is more favorable where concerns about biosolids odour and
221 the occurrence of pathogenic agents in biosolids outweigh those about air pollution.
222 Regardless of the final disposal options, the efficiency of the dewatering process is important
223 for cost-effective management of the biosolids since a high solid content can reduce the cost
224 of transportation for land application and consumption of auxiliary fuel for incineration.

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

233 Biosolids odour can be a major roadblock for beneficial reuse of the solid residuals from
234 anaerobic digestion. These odorants primarily include six types of volatile compounds
235 including sulfur bearing compounds, nitrogen bearing compounds, VFAs, ketones, aldehydes,
236 and hydrocarbons^{31,32}. Most studies to date have focused on sulfur bearing compounds when
237 assessing biosolids odour emissions since they are the primary constituents of odour from
238 biosolids following anaerobic digestion of protein-rich substrates^{33,34}. These studies have
239 provided important insights to connect the issue of biosolids odour and AcoD with sulfur rich

240 co-substrates ³⁵. Of a particular note, Higgins et al., ³⁶ proposed a cyclic pathway to describe
241 the production and transformation of volatile sulfur compounds and H₂S. They hypothesize
242 that the addition of sulfur rich co-substrates during AcoD can result in an increase in
243 biosolids odorous emission. This involves processes such as degradation of protein,
244 generation of associated volatile organic sulfur compounds (e.g. methanethiol) and
245 subsequent formation of H₂S ^{4, 36, 37}.

246 The sensorially relevant volatile compounds are predominantly generated from microbial
247 degradation of organic matter ^{38, 39}. p-cresol, trimethylamine and VFAs have been
248 demonstrated as key sensorially relevant volatile compounds that are associated with
249 anaerobic degradation of organic matter ^{31, 39}. In addition, a range of aromatic compounds
250 (e.g. toluene, p-cresol and indole) can be generated through anaerobic degradation ^{32, 40}.
251 Although detected at low levels (i.e. <500 µg/m³), these aromatic compounds are considered
252 to be important odorants when peaks of volatile sulfur compounds diminish ⁴⁰.

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

262 3. Addressing the impact of AcoD on downstream processes

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

268 [FIGURE 3]

269 3.1. Biogas quality, digestate dewaterability and biosolids odorous emissions

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

274 3.1.1. Co-substrate selection

275 The variations in co-substrate properties and composition govern the AcoD process
276 (particularly biogas quality), and thus the downstream processes. In some cases, co-digestion
277 can lead to synergistic effects, reflected by either a boost in specific methane yield or an
278 increase in biogas production kinetics^{4, 46}. The synergistic effects are often associated with
279 higher volatile solid removals and improved biogas quality (e.g. higher methane content),
280 likely associated with the balanced C/N ratio and dilution of inhibitory and toxic substances
281^{47, 48}. The extent of the synergistic effects can be notable during AcoD⁴⁷⁻⁵¹. In contrast,
282 antagonistic effects have also been observed during AcoD of sewage sludge and organic
283 waste⁵². Antagonistic effects have negative impact on overall downstream processes. It is
284 noteworthy that the types of carbon source from metabolic degradation of co-substrates

285 during AcoD may also affect methanogenic activities and pathways, thus leading to varied
286 biogas quality ⁵³.

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

298 Coagulants (e.g. aluminum or ferric salts) addition to sewage sludge can reduce the
299 generation of total volatile organic sulphur compounds ^{55, 56}. Indeed, these coagulants can
300 react with sulphide to form precipitate. Thus, metal addition (e.g. iron and aluminum) can be
301 used for hydrogen sulfide control, and subsequently total volatile organic sulphur compounds
302 control, as total volatile organic sulphur compounds can be generated by microbial
303 conversion of hydrogen sulfide to methanethiol ^{36, 57}. It is noteworthy that by supplementing
304 iron (e.g. Fe⁰, Fe(II) and Fe(III)) to sewage sludge during AcoD, the release of bound
305 proteins for further biodegradation can be expected due to iron's action as electron
306 donor/acceptor and micronutrient (i.e. cofactor of key enzymatic activities) ^{58, 59}. Thus, there
307 is possible increase in volatile solid destruction and process performance at the presence of
308 iron addition during AcoD. Indeed, Vrieze et al. ⁶⁰ observed more stable methane production

309 during AcoD of kitchen waste and sludge amended with Fe(II), compared to mono-digestion
310 of kitchen waste.

311 3.1.2. Co-substrate pre-treatment

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

316 Thermal pretreatment can increase the final solid content of biosolids, possibly due to the
317 reduction in the bound water in digestate and thus viscosity of the digestate ⁶⁷. At a higher
318 thermal pretreatment temperature, a greater solubilization of organics occurs. This allows the
319 remaining materials to be smaller in size and less amenable to removal by polymer
320 conditioning, as polymer conditioning can mainly remove biocolloids in the range of up to
321 1.5 μm ^{61, 68}. Studies have been performed to evaluate the effect of thermal hydrolysis
322 pretreatment temperature on subsequent digestion performance and operation, as well as
323 downstream parameters such as dewatering and cake quality ^{61, 63}. The authors found that
324 higher thermal hydrolysis temperatures improved the volatile solid reduction and biogas
325 quality, as well as cake solids after dewatering ⁶¹. In addition, thermally pretreated co-
326 substrates (e.g. microalgae) can facilitate the release of nutrients during AcoD with sewage
327 sludge ⁶⁹. Other pretreatment methods such as thermo-oxidative technique can also enhance
328 the removal of volatile sulfur compounds and lead to optimised AcoD processes and
329 mitigation in biosolids odorous emissions ⁷⁰.

330 Chemical pretreatment can have additional effects on facilitating sludge dewatering and
331 biogas production when combined with thermal pretreatment ⁷¹. Multivalent cations (such as
332 Ca^{2+}) exhibit positive effects on improving sludge dewaterability by forming hydroxybases or

333 changing digestate/sludge properties as surface charge, viscosity and floc strength ^{72, 73}.
334 Recently, Song et al. ⁶⁵ applied a combination of persulfate and zero valent iron to enhance
335 the dewaterability of anaerobically digested sludge. Chemical pretreatments can also be used
336 to facilitate downstream nutrient recovery. For example, Shi et al. ⁷⁴ used acidified pig
337 manure to recover nitrogen, phosphorus and VFAs using bipolar membrane electrodialysis⁷⁴.
338 Biological pretreatment is a promising technique to improve biosolids dewaterability,
339 although this has not been applied at full scale. Murugesan et al. ⁷⁵ used an *Acidithiobacillus*
340 *ferrooxidans* culture to pre-condition the digestate and observed a noticeable decrease in
341 extractable EPS content. They suggested a possible application via bioacidification treatment
342 to improve sludge dewaterability ⁷⁵.

343 3.1.3. Process optimisation

344 Operating parameters including temperature, retention time and organic loading can be
345 optimized to tackle the negative downstream processes. For example, dewaterability of
346 mesophilic digestate is usually better than that of thermophilic digestate due to better process
347 stability at mesophilic conditions ⁷⁶. On the other hand, an optimised AcoD process with
348 adequate solid retention time and organic loading can enhance volatile solid destruction. This
349 leads to an enhanced VFA yield and specific methane yield, thus reducing biosolids odorous
350 emissions. As VFA concentrations can be empirically corrected with the odor production
351 potential ³⁸, not only is the enhanced VFA production during AcoD process beneficial for
352 subsequent VFAs recovery or methanogenesis processes, but it also is pivotal in reducing
353 biosolids odorous emissions ⁷⁷. In some instances, it can be achieved by coupling the AcoD
354 process with a bioelectrochemical systems (e.g. microbial electrolysis cell) to simultaneously
355 recover VFAs and ammonia, as well as avoid AcoD system instability ⁷⁸. In addition,
356 optimised process design can improve digestate dewaterability. For example, Cobble Dick et

357 al. ⁷⁹ explored the possibility of applying recuperative thickening technology for enhancing
358 biogas production and dewaterability in anaerobic digestion processes. It is noteworthy that
359 recuperative thickening technology can be employed in conjunction with AcoD to enhance
360 volatile solid destruction and the methane yield ⁷⁹.

361 3.1.4. Biological desulphurization

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

377 3.2. Nutrient management

378 There have been several excellent reviews on nutrient (particularly phosphorus) recovery
379 from digestate ^{7, 30, 83-85}. Technologies for phosphorus removal and recovery can be broadly

380 divided into four major categories, including chemical precipitation, enhanced biological
381 treatment, crystallisation, and membrane based technologies (Table 2).

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

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

404 Crystallisation technologies have been applied at several full scale plants to recover P in the
405 form of struvite ($\text{MgNH}_4\text{PO}_4 \cdot 6\text{H}_2\text{O}$), and to a lesser extent, hydroxyapatite ($\text{Ca}_{10}(\text{PO}_4)_6(\text{OH})_2$)
406 or calcium phosphate ($\text{Ca}_3(\text{PO}_4)_2$)⁹⁰⁻⁹². The key driver for phosphorus recovery in these
407 plants is to prevent uncontrolled struvite scaling or struvite blockage within the plant^{90,91}. It
408 is noteworthy that, at the present, revenue from struvite recovery (given the current market
409 value of phosphorus fertilizer) is not sufficient to recover the cost of this process. Thus,
410 further optimisation is necessary to improve the efficiency and economic viability of
411 phosphorus recovery if crystallization technologies are adapted for AcoD applications. The
412 co-precipitation of calcium, ferric, aluminium^{93, 94} and organic compounds^{21, 95, 96}
413 significantly reduces the recovery and purity of struvite and inhibits the crystal growth due to
414 the confined space where crystals could be formed^{97, 98}. Hence, liquid matrix can undergo
415 some pre-treatment (e.g. acid leaching, chelating agents and microwave treatment) to
416 minimise the inhibitory effect prior to phosphorus recovery via struvite precipitation^{99, 100}.

417 The efficiency of crystallisation for phosphorus recovery is directly proportional to the
418 concentration of phosphorus in the sludge centrate. For example, due to competition from
419 impurities in the centrate, the chemical cost for phosphorus recovery increases exponentially
420 as the phosphorus concentration in the initial feedstream decreases. Thus, innovative
421 solutions, such as membrane based technologies, are needed to improve the economic
422 feasibility of struvite precipitation process (Table 2). Membrane based technologies, in
423 particular forward osmosis and electrodialysis, have the potential to simultaneously produce
424 high quality effluent and pre-concentrated wastewater to facilitate nutrient recovery^{7, 101}. For
425 example, Xie et al.^{20, 102} developed a novel forward osmosis - membrane distillation hybrid
426 system for extracting phosphorus from sludge centrate in the form of struvite. Forward
427 osmosis can concentrate orthophosphate and ammonium for subsequent phosphorus recovery
428 with higher crystal growth kinetics. On the other hand, Shi et al.⁷⁴ used bipolar membrane

429 electro dialysis to recover ammonium, phosphate and VFAs simultaneously from both
430 synthetic and real pig manure hydrolysate.

431 [TABLE 2]

432 4. Conclusion

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

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447 6. References

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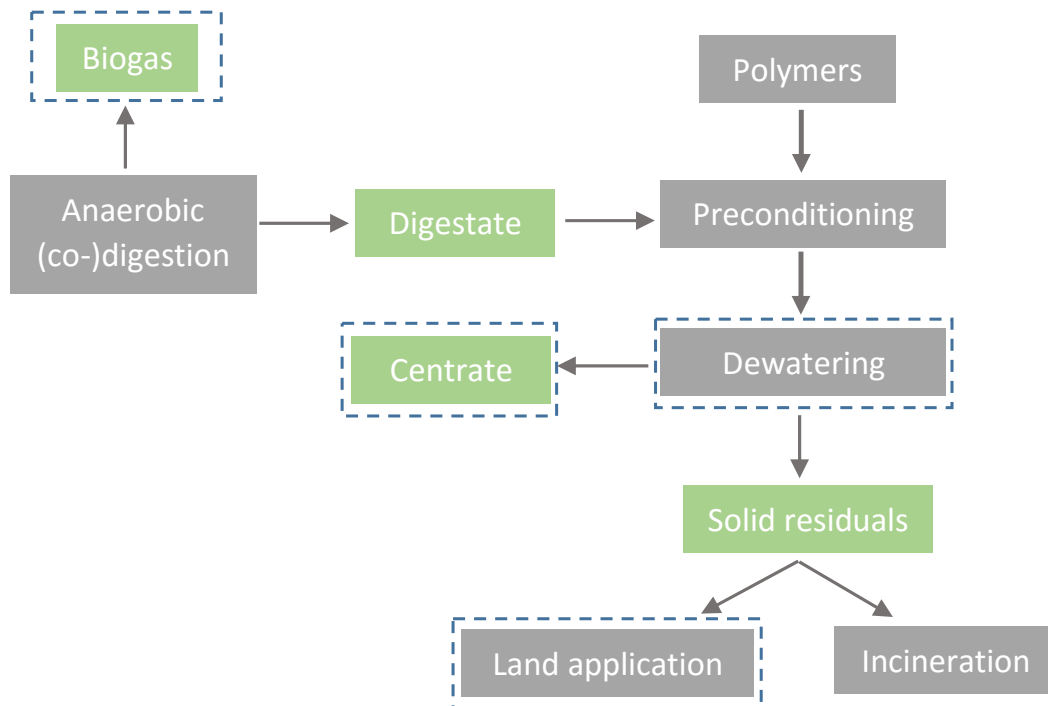
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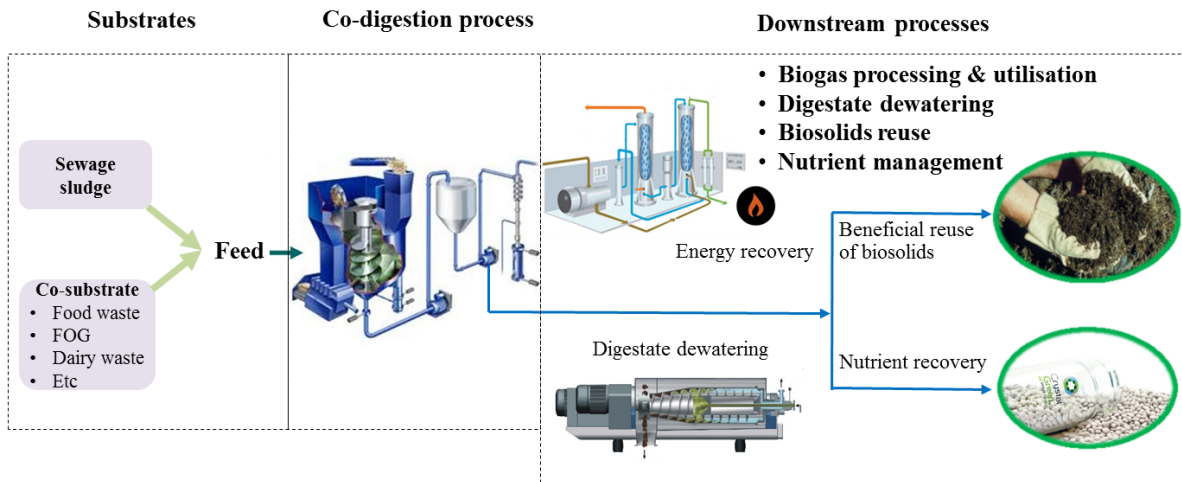
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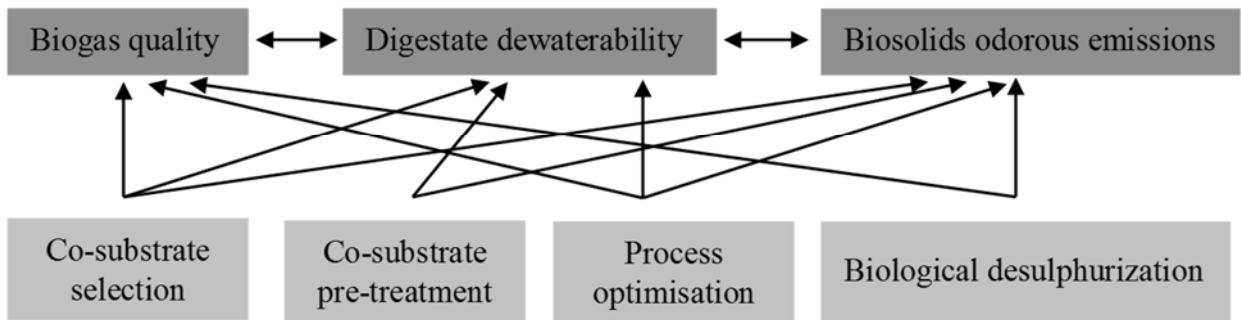
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741 **Figure 1:** A schematic diagram for anaerobic digestion associated downstream processes
 742 shown in dashed squares: (1) digestate dewaterability; (2) biosolids odorous emissions; (3)
 743 nutrient management; and (4) biogas quality. Green-filled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



745

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



747

748 **Figure 3:** Perspectives on strategies to improve downstream processes: biogas quality,
 749 digestate, dewaterability, and biosolids odorous emissions. Correlated processes are denoted
 750 as \longleftrightarrow .

751 **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

752

753 **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

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