Beyond land application: Emerging technologies for the treatment and reuse of anaerobically digested agricultural and food waste

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Abstract

Effective treatment and reuse of the massive quantities of agricultural and food wastes generated daily has the potential to improve the sustainability of food production systems. Anaerobic digestion (AD) is used throughout the world as a waste treatment process to convert organic waste into two main products: biogas and nutrient-rich digestate, called AD effluent. Biogas can be used as a source of renewable energy or transportation fuels, while AD effluent is traditionally applied to land as a soil amendment. However, there are economic and environmental concerns that limit widespread land application, which may lead to underutilization of AD for the treatment of agricultural and food wastes. To combat these constraints, existing and novel methods have emerged to treat or reuse AD effluent. The objective of this review is to analyze several of the emerging methods used for efficient treatment and reuse of AD effluent. Overall, the application of emerging techniques is limited by AD effluent composition, especially the total solids content. Some technologies such as composting, use the solid fraction of AD effluent, while most other technologies, such as algae culture and struvite crystallization, use the liquid fraction. Therefore, dewatering of AD effluent, reuse of the liquid and solid fractions, and land application could all be combined to sustainably manage the large quantities of AD effluent produced. Issues such as pathogen regrowth and prevalence of emerging organic micro-pollutants are also discussed.

Keywords: anaerobic digestion, land application, AD effluent, resource recovery, food waste, agricultural waste

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1. Introduction

In the United States, more than 300 million tons of organic wastes are generated each year from agricultural and food processing industries [1, 2]. Sustainable treatment of these huge amounts of organic wastes has become a global challenge. Anaerobic digestion (AD) has been used throughout the world to stabilize food wastes, manure and other organic wastes through volatile solids (VS) mineralization and pathogen reduction [3]. Additionally, AD has gained recognition for its added benefit of producing renewable energy. During the AD process, anaerobic microorganisms convert organic matter into biogas and digestate (called AD effluent). The biogas, made up of mainly methane (~60%) and carbon dioxide (~40%), can be used to produce heat, electricity, compressed natural gas (CNG), and/or liquefied natural gas (LNG) [4]. Meanwhile, the nutrient-rich AD effluent is commonly used as an organic fertilizer [5]. Recently, increasing attention has been paid to AD due to its capability to produce bioenergy from a broad range of feedstocks, including agricultural residues and forestry wastes [3].

In the U.S., there are 247 AD systems in operation at commercial livestock facilities [6]. It is estimated that 8,200 livestock facilities could benefit from AD [7]. There are even fewer AD systems dedicated for food waste, which contributes over 12% of the organic waste produced in the U.S [1, 8]. One reason for underutilization of AD could be the low price of electricity, which reduces the economic incentive to use AD for energy production. For example, Yiridoe et al. [9] estimated that large livestock operations (600 dairy cows or 800 sows) are required to make an on-farm AD system economically feasible. The expanding world population will increase the demand for agricultural and food products, causing increased quantities of organic wastes, increasing the potential for AD application and, consequentially, the quantities of AD effluent. The difficulty in treating and reusing large quantities of AD effluent is another likely cause of underutilization of AD technology [10]. The most common AD effluent treatment is land application for crop production, because large quantities can be spread over large areas of farmland [5]. However, improper land application practices can lead to ammonia volatilization, pathogen contamination, over-fertilization and nutrient runoff, sparking considerable

debate about the environmental sustainability of this practice [5]. To improve economic and environmental sustainability of AD, alternative methods for AD effluent management are desirable.

Many existing technologies, such as ammonia stripping, composting, reverse osmosis and struvite crystallization, have been used for treatment and reuse of other wastes, including digested sewage sludge and municipal wastewater [11, 12]. Recently, these techniques have emerged as effective methods for high throughput treatment of AD effluent [13-16]. Many emerging methods (algae production, hydroponics, freshwater source for ethanol production) can reuse the water and nutrients (nitrogen and phosphorus) present in the liquid fraction of AD effluent [17-19]. Additionally, fibers in the solid fraction can be effectively reused for the production of fuels, particleboard, and bedding materials [20-22].

Integration of emerging technologies with responsible land application practices could improve the economic feasibility and environmental sustainability for the AD of food and agricultural wastes. The objective of this study was to analyze emerging solutions for management of the water, solids, organic matter, and nutrients in AD effluent. Effluent from anaerobic digesters used for municipal wastewater treatment is outside of the scope of this review. However, many management techniques developed for municipal anaerobic digester effluent, such as composting and struvite crystallization, can be applied to food waste/agricultural waste AD effluent, and are summarized in the designated sections of this review.

Figure 1 outlines a proposed process for the reuse and treatment of AD effluent using currently available technologies. In this process, a portion of AD effluent is land applied for crop production. Meanwhile, other portions are reused as AD inoculum or dewatered to produce a liquid and solid fraction. The liquid fraction of AD effluent may be used as a freshwater source for ethanol production. Concurrently, a portion of the liquid fraction can be sent to microbial fuel cells, which degrade residual organics and produce sustainable electricity. Meanwhile, nitrogen in the liquid fraction can be removed by ANAMMOX and/or recovered as ammonium sulfate $((\text{NH}_4)_2\text{SO}_4)$ by ammonia stripping and absorption. Additionally, both the nitrogen and phosphorus in the liquid fraction can be used for algae/hydroponic plant production or converted to struvite, a renewable magnesium ammonium phosphate fertilizer. The solid fraction of AD effluent can be composted with lignocellulosic biomass, converted into

Waste stream*

Figure 1: Proposed processes for beneficial reuse and treatment of AD effluent

2. Framework for AD effluent management

2.1 Composition

A recent compositional analysis of AD effluent used for agronomic land application is summarized in Table 1. The composition of AD effluent is mainly dependent on feedstock components and AD operating conditions (e.g. temperature, retention time). AD effluent is usually analyzed for dry matter (% DM) or total solids (% TS); organic matter (% OM) or volatile solids (% VS); total carbon (TC); total nitrogen (TN); total ammonia nitrogen (TAN); total phosphorus (TP); total potassium (TK); sulfur (S);, minor nutrients, such as magnesium (Mg) and calcium (Ca); heavy metals (As, Cd, Cr, Cu, Hg, Mn, Ni, Pb, Zn); pathogens (commonly using fecal coliforms as an indicator); chemical oxygen demand (COD); and biochemical oxygen demand (BOD) [23].

Table 1: AD effluent composition from digesters feeding food waste or agricultural feedstock

Component and unit	Feedstock			
	Food Waste ^a	Reference(s)	Agricultural feedstock ^b	Reference(s)
pH	7.9-8.2	[24-27]	7.5-8.4	[28-31]
DM (%)	1.4-6.1	[24-27, 32, 33]	7.4-24.0	[28-31, 34]
VS (% of DM)	38-71	[24, 26, 32, 33]	69-74	[29, 35]
TC (g kg ⁻¹ DM)	311-452	[24-26]	300-400	[28-31]
TN (g kg ⁻¹ DM)	112-157	[24-27, 32, 33]	22-88	[28-31, 34, 35]
TAN (g kg ⁻¹ DM)	52-108	[24-27, 32]	6-45	[28-31, 34, 35]
TP (g kg ⁻¹ DM)	6-16	[24, 25, 27, 32, 33]	2-66	[28, 30, 31, 34, 35]
TK (g kg ⁻¹ DM)	26-81	[24, 25, 27, 32, 33]	9-100	[28, 30, 31, 34, 35]
S (g kg ⁻¹ DM)	6-10	[24, 25, 27]	-	NR
Mg (g kg ⁻¹ DM)	3-10	[24, 25, 27]	-	NR
Ca (g kg ⁻¹ DM)	10-52	[25, 27]	31	[31]
BOD (mg L ⁻¹)	NR	-	5900	[29, 30]
COD (mg L ⁻¹)	NR	-	5600	[34]
As (mg kg ⁻¹ DM)	NR	-	29	[36]
Cd (mg kg ⁻¹ DM)	0.3-1.0	[25, 32]	0-10	[30, 36, 37]
Cr (mg kg ⁻¹ DM)	13-29	[25, 32]	2-55	[30, 36, 37]
Cu (mg kg ⁻¹ DM)	14-70	[25, 27, 32]	1-1373	[31, 36, 37]
Hg (mg kg ⁻¹ DM)	<0.1	[25]	ND	[37]
Mn (mg kg ⁻¹ DM)	0-201	[25, 27]	NR	-
Ni (mg kg ⁻¹ DM)	25-39	[25, 32]	2-36	[30, 36, 37]
Pb (mg kg ⁻¹ DM)	0.1-10	[25, 32]	0-46	[30, 36, 37]
Zn (mg kg ⁻¹ DM)	113-474	[25, 27, 32]	43-3884	[31, 36, 37]

^afood wastes include food slurry, slaughterhouse wastes, source separated household wastes, and kitchen and garden wastes; ^bagricultural feedstock includes manures, silage, and crop residues; NR=not reported; ND=not detected

AD operations and feedstock characteristics clearly impact the DM content, with values between 1% and 24% (Table 1); however, typical DM contents of AD effluent produced during commercial AD of animal manures and food wastes ranged from 2-9% [25, 28, 30]. During the AD process, organic nitrogen in feedstock is degraded, but only very little is assimilated by AD microorganisms [3]. As a result, AD effluent usually contains high levels of TN (up to 157 g TN kg⁻¹ DM) which is mostly TAN (50-99%), a combination of ammonium (NH₄⁺) and free ammonia (NH₃) [28]. The ratio of NH₄⁺ to NH₃ varies with pH and temperature according to ion dissociation balance. The carbon to nitrogen (C/N) ratio is usually used as an indicator to assess the applicability of AD effluent as an amendment to a specific soil type. The change in C/N ratio is dependent on feedstock carbon degradability, because the nitrogen content is relatively stable in AD [38]. The phosphorus content comes from adenylates, nucleic acids and phospholipids, which are in the feedstocks [38]. The pH (7.5-8.0) in AD effluent increases the solubility of phosphate (PO₄³⁻), causing precipitation into calcium or magnesium phosphates (Ca₃(PO₄)₂ or Mg₃(PO₄)₂) and struvite, a complex crystal that contains nitrogen, phosphorus, and magnesium (Section 4.3.4) [38]. The sulfur (S) enters digesters from feedstock proteins, and over supply can lead to hydrogen sulfide (H₂S) production, which negatively affects biogas quality [38, 39]. Heavy metals in AD effluent have been predicted to originate in commercial animal feeds and AD seeding sludge [37]. However, the concentrations of heavy metals in Table 1 are well below the ceiling concentration limits for land application set by the U.S EPA [40]. None of the studies measured pathogen levels in AD effluent prior to land application. To ensure protection of public and animal health, future studies should also monitor pathogens prior to land application [41].

As mentioned earlier, feedstock variability and digester operating parameters affect the composition of AD effluent, making generalized statements about composition very difficult [5]. For example, mesophilic AD (37°C) of feedstocks with high levels of TN and degradable organics produces AD effluent with high TAN and a low C/N ratio [38]. Additionally, thermophilic AD (55°C) can increase pathogen and VS destruction [42], producing effluent with safer amendment properties and lower C/N

ratios, respectively. Still, the nitrogen to phosphorus (N/P) ratio is relatively stable in AD effluent [38].

This fact has prompted the use of AD effluent as an organic fertilizer for the production of crops.

2.2 Land application for crop production

As shown in Table 2, several crops including maize, silage [31], ryegrass [28], legumes [43], and fruits [44] have been successfully grown using AD effluent as an organic fertilizer. The local climate, physical and biological soil properties, AD effluent composition, nitrogen application rates, plant-available nitrogen, crop type, and crop rotations substantially affected plant yield and nitrogen uptake. For example, Sieling et al. [45] studied the nitrogen balance during a three year crop rotation using digested manure and raw manure as organic fertilizers as well as mineral fertilizers. Overall, grass yields and nitrogen uptake were lower when digested manure was used, but there was higher nitrogen balance (difference between nitrogen fertilization and plant nitrogen uptake) than for the mineral fertilizer or raw manure [45]. Svensson et al. [46] compared land application of compost and AD effluent, both from source-separated domestic wastes, and concluded that the composts needed nitrogen supplementation while AD effluent needed phosphorus supplementation in order to maintain grain quality and yields [46].

Generally, crop yields and nitrogen uptake for experiments conducted in pots have been similar using AD effluents, organic fertilizers, and inorganic fertilizers. For example, Walsh et al. [47] showed that grass yields were comparable in 1.2-L pots using liquid AD effluent, mineral fertilizers, and other organic fertilizers. This was attributed to the broad nutrient base within AD effluent [47]. Haraldsen et al. [27] showed that barley production and nutrient uptake rates were comparable between household waste AD effluent and inorganic fertilizer for applications with equivalent amounts of nitrogen. Bougnom et al. [43] showed that the combination of AD effluent and ash was a good soil amendment for grass, herbs, and leguminous plant production, because the amendment did not significantly affect the acidic soil's physical and chemical properties [43]. Abubaker et al. [25] showed that wheat yields were similar when using AD effluent or mineral fertilizers. An additional advantage was that the AD effluent actually increased the nitrogen mineralization capacity and partial ammonia oxidation rate in the soil [25]. Fouda et al. [28] tested several AD effluents and showed that ryegrass efficiently utilized nitrogen (80-100%) for all of

1 them. The nitrogen balance was correlated to the amount of organic nitrogen applied, and the authors
2 concluded the organic C/organic N ratio should be low to maintain good short-term nitrogen availability
3 [28]. Similarly, phosphorus uptake by maize (*Zea mays* L.) was comparable when Bachmann et al. [48]
4 used digested dairy slurry and maize silage, raw dairy slurry, or mineral fertilizers. Interestingly, Yu et al.
5 [44] grew tomatoes in AD effluent, concentrated AD effluent, and traditional fertilizers. Their results
6 showed that 16 kinds of amino acids, protein, beta-carotene, soluble sugar, vitamin C, and tannins
7 increased after application of concentrated AD effluent.

8 Field scale results have also been promising. In a three-year study, Gagnon et al. [31] land
9 applied digested swine manure at 146-152 kg nitrogen ha⁻¹, resulting in application of 73-117 kg
10 phosphorus ha⁻¹. Similar corn silage yields were observed with either digested manure or undigested
11 manure. However, crop nitrogen utilization from the organic fertilizers was only 70% as efficient as crops
12 using mineral nitrate fertilizers [31]. Loria et al. [49] showed that digested and raw swine manure were
13 not significantly different as a source of nitrogen for growing corn. This result was attributed to similar
14 grain yields and plant nitrogen uptake, despite variable weather conditions. In another example, Möller et
15 al. [50] showed that yields of several crops were not significantly different between raw and digested
16 animal slurry. Additionally, Chantigny et al. [51] concluded that digested liquid swine manure had similar
17 fertilizer value to mineral fertilizers based on similar grain yields (6-11 t ha⁻¹), nitrogen uptake (83-176 kg
18 ha⁻¹), and phosphorus uptake (19-40 kg ha⁻¹), and attributed the results to land application practices that
19 limited ammonia volatilization (Section 2.3). All this evidence points to the fact that AD effluent can be
20 applied as an effective organic fertilizer for agricultural crop growth. However, no studies have examined
21 the effects of AD operational parameters on AD effluent composition, studied the long term effects of AD
22 effluent on physical and biological soil properties, nor examined nutrient mass balance and energy
23 balances at field scale levels. Evaluation of these factors could provide better understanding of the
24 impacts of land application [5]. There are several additional concerns that must be addressed, including
25 pathogen exposure, nutrient runoff, and transport economics. These issues have limited the widespread
26 adoption of AD effluent land application.

Table 2: Crop production using AD effluent as nitrogen fertilizer

AD feedstock	Application Rate (kg N ha ⁻¹)	Crop	Time Period	Plot Area	DM Yield	N uptake (kg N ha ⁻¹)	Reference
Dairy slurry	100,50 ^a	Ryegrass	16 wk	0.018 m ² pot ⁻¹	4-6 g pot ⁻¹	NR	[47]
Dairy slurry	100,50 ^a	Ryegrass/clover	16 wk	0.018 m ² pot ⁻¹	4-8 g pot ⁻¹	NR	[47]
Slaughterhouse waste	35-140 ^b	Wheat	77 d	0.020 m ² pot ⁻¹	5-25 g pot ⁻¹	NR	[25]
Dairy slurry/maize silage/wheat grain	1.13 ^c	Maize	8 wk	0.035 m ² pot ⁻¹	59-65 g pot ⁻¹	0.451-0.512 ^c	[48]
Household waste	80,160	Barley	103 d	0.036 m ² pot ⁻¹	18, 21 g grain pot ⁻¹	0.18,0.28 ^c	[27]
Maize silage/cattle manure	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.3-1.6 ^c	[28]
Maize silage/poultry manure/unidentified slurry	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.5-1.6 ^c	[28]
Maize silage/wheat silage/dairy manure/swine manure/other	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.5-1.6 ^c	[28]
Maize silage/cattle manure/other	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.5-1.6 ^c	[28]
Maize silage/wheat silage/other	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.2-1.6 ^c	[28]
Maize silage/clover, ryegrass silage/other	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.4-1.6 ^c	[28]
Maize silage/wheat silage/other	0.3 ^c	Ryegrass	309 d	0.05 m ² pot ⁻¹	NR	1.5-1.6 ^c	[28]
Cattle manure/swine manure	2.56 ^c	Tomato	5 mo	12 m ²	121 ^d	NR	[44]
NR	190	Herbs/grasses/legumes	4 mo	20 m ²	4-4.5 ^e kg	NR	[43]
Swine manure	146-152	Silage Corn	3 yr	21 m ²	7-13 t ha ⁻¹	54-99	[31]
Swine manure	100	Grain corn	3 yr	36 m ²	6-11 t ha ⁻¹	83-176	[51]
		Clover/winter					
Cattle manure	0-308	wheat/potatoes/maize/rye/peas/spelt/spring wheat/grass	3 yr	72 m ²	118 t ha ⁻¹	194	[50]
Household waste/food residues	100	Oats/barley	4 yr	90 m ²	2-4 t ha ⁻¹	39-77	[46]
Swine manure/maize silage	0-450	Clover grass/Perennial Ryegrass	2 yr	144 m ²	11.6 t ha ⁻¹	342	[45]
Swine manure	76-172	Grain corn	3 yr	835 m ²	4-12 t ha ⁻¹	100-250	[49]
Animal manure	130-150	Energy maize	2 yr	0.8 ha	NR	NR	[52]

^atwo fertilization rates; ^bNH₄⁺-N; ^cg N pot⁻¹; ^dfruit weight; ^efresh matter; NR=not reported

2.3 Concerns with land application

As mentioned previously, heavy metals are not a significant concern in AD effluent produced from agricultural and food wastes. However, several pathogenic organisms, including *Salmonella*, *E. coli*, and *Clostridia*, may be present in these AD feedstocks [41]. The inactivation of pathogens in AD is dependent on the initial amount of pathogens in feedstocks, digestion parameters (temperature, time), and nutrient contents [41]. Temperature is the most important factor affecting survival of pathogens in AD [41]. Most AD facilities operate under mesophilic conditions (37°C), due to better process control and lower heating requirements [42]. However, pathogens survive more readily during mesophilic digestion [41].

The pathogens in mesophilic AD effluent may be a significant bottleneck to widespread acceptance of land application of AD effluent. For example, AD effluent with high pathogen levels could cause public health concerns, potentially reducing the land area available for agronomic application [41]. Food safety has become a hallmark of U.S. regulations, which may influence farmer perception of using organic fertilizers for crops for human consumption. Consequently, many large concentrated animal feeding operations (CAFOs) with AD operations may have limited outlets for AD effluent disposal [53].

Pathogens can be reduced through thermophilic AD (55°C) or other conditioning methods, including composting, heat drying, heat treatment, beta-ray or gamma-ray irradiation, and pasteurization [11]. However, many of these methods are costly, which has made lagoon storage an economical alternative to increase available storage to allow more flexibility in the timing of land application [54]. Out of these methods, composting has become a popular treatment method for AD effluent produced from agricultural and food wastes. This is primarily because of the operational simplicity and low costs associated with composting (Section 5.1). However, the high moisture content of AD effluent often requires addition of bulking materials, such as sawdust.

Irresponsible management of AD effluent nutrients also poses significant concerns. Traditional surface application methods, such as spray irrigation can cause volatile emissions (e.g. NH₃). This is especially true for the liquid fraction of AD effluent after dewatering (Section 3.2), which contains higher

proportions of the volatile TAN [5]. Depending on the soil microbial activity, nitrification and denitrification of NH_4^+ to nitrous oxide (N_2O) can cause negative impacts on local air quality [55]. Over-supply of AD effluent can cause nutrient runoff [25]. For example, the nitrogen to phosphorus (N/P) ratio in AD effluent is relatively stable. Application of AD effluent with low a N/P ratio for production of crops that require a high N/P ratio may cause over-application of phosphorus, increasing the probability of runoff and eutrophication in nearby waters. Additionally, the stable N/P ratio limits the ability to control nutrient composition during farming, which may reduce farmer acceptance. Furthermore, concerns over NO_3^- emissions to surface water and groundwater may lead to regulations that limit the quantity of AD effluent allowed to be land applied [5]. Finally, long transportation distances reduce the economic viability of AD effluent for land application [56]. Without close proximity to agricultural fields, it may be more cost effective to use incineration, landfilling, composting or lagoons.

The issues of ammonia volatilization and nutrient runoff can be resolved using direct injection methods for reduced air-AD effluent contact [57] and springtime land application to exploit higher nitrogen uptake rates by plants [11]. Advanced nutrient recovery techniques, such as struvite crystallization (Section 4.3.4), may be used to control nutrient composition in AD effluent. However, the diversity of environmental and economic concerns has prompted significant debate about the sustainability of land application of AD effluent, which has been thoroughly discussed in previously published reviews [5, 38]. Additionally, there are limited long term studies on the effects of land application on nutrient runoff, volatile emissions, pathogens, and economics [5]. The adoption of dewatering practices (Section 3.2) has necessitated agronomic studies on using the liquid or solid fraction of AD effluent as fertilizers. These issues have prompted development of emerging technologies to reuse, remove, and recover components of AD effluent.

3. Emerging techniques for treatment and reuse of raw AD effluent

One of the main advantages of land application is that AD effluent can be recycled directly after the AD process. However, there are only a few emerging options for raw AD effluent management, which

are summarized below. Most of the emerging methods use either the liquid or solid fractions of AD effluent after dewatering (Section 4, Section 5).

3.1 Reuse as AD inoculum

AD effluent is rich in active anaerobic microbes and nutrients, and can be re-used as AD inoculum. If high solids feedstocks, such as lignocellulosic biomass, are used for digestion, raw AD effluent can provide water to obtain an optimal DM content (20–25%) for solid state anaerobic digestion (SS-AD). AD effluent can be used as inoculum directly through recirculation to AD reactors, or indirectly by mixing with feedstocks.. The inoculation procedure is different for L-AD and SS-AD reactors. To apply AD effluent as L-AD inoculum, effluent and feedstock are usually mixed and agitated inside the digester. This is often conducted by partial re-circulation. To use AD effluent as inoculum for SS-AD , effluent and feedstock need to be premixed outside of the digester, due to reduced ability to agitate mixtures with high solids contents. Partial premixing can be used to reduce energy demands, and has achieved similar methane yields compared to complete premixing [58]. Other methods, such as injection of inoculum into SS-AD reactors, may reduce inoculation size and energy costs. Produced biogas can be converted to heat using a combined heat and power (CHP) unit to maintain the temperature of both L-AD and SS-AD reactors [58]. The digestate from SS-AD has a higher DM content compared to L-AD effluent and can be land applied or composted.

L-AD effluent is more favorable than other sources of inocula (activated sludge, manure, rumen fluid), because it contains large populations of active methanogens and sufficient and available nutrients, and most importantly, it has been acclimated to the AD environment. There are a few studies that have compared the performance of different inocula sources for the production of methane by AD (Table 3). The available literature showed that AD effluent can initiate and produce relatively higher methane yields than alternative sources at both mesophilic and thermophilic temperatures. In one study, six inoculum types were employed for thermophilic SS-AD of the organic fraction of municipal solid wastes (OFMSW) [59]. Results showed that AD effluent had the highest methane yield of 530 L/kg-VS and the

shortest AD lag phase. Performance of effluent recirculation was also tested, and a 16% increase in the methane yield was achieved when effluent was recirculated, compared to water-feedstock dilution (1:1 dilution ratio) [60]. A recent study utilized autoclaved effluent to evaluate effects of microbial and non-microbial factors on inoculum performance [61]. The results showed that non-microbial factors, such as alkalinity, were most influential on methane yield for SS-AD at feedstock to effluent (F/E, VS/VS (g/g)) ratios of 2.2–6.6.

Table 3: Comparison of different inocula for anaerobic digestion

Feedstock	TS (%)	Temperature (°C)	Inoculum sources	CH ₄ yield (L kg ⁻¹ -VS)	Lag phase (# of days)	Ref.
Rice straw	-	37	paper mill sludge	10	-	[62]
			digested municipal sludge	15	-	
			digested chicken manure	65	8	
			anaerobic granular sludge	110	10	
			digested swine manure	150	10	
			digested dairy manure	175	7	
Animal byproducts	2.0	35	UASB effluent	195	21	[63]
			wastewater sludge	317	10	
			anaerobic lagoon sludge	1000	31	
			landfill leachate	1059	20	
Corn stover	22	37	municipal sewage sludge AD effluent	120	2	[64]
			food waste AD effluent	195	2	
			dairy waste AD effluent	238.5	2	
OFMSW	30	55	cattle excrement	30	2-4	[59]
			corn silage	110	2-4	
			swine excrement	180	2-4	
			restaurant waste digested with rice hulls	220	2-4	
			swine mixed with digested sludge	270	2-4	
			digested sludge	530	2-4	

OFMSW: organic fraction of municipal solid wastes; UASB: Up-flow anaerobic sludge blanket reactor

One possible concern of recirculating AD effluent is the accumulation of TAN, volatile fatty acids (VFAs), and other metabolic products that may inhibit microbial activities. Therefore, AD effluent recirculation needs to be coupled with fresh water dilution or pretreated to control inhibitors. For instance, Nordberg et al. [65] reported that effluent recirculation resulted in increased pH and alkalinity during AD of alfalfa silage, which made it possible to increase the organic loading rate, but eventually caused

inhibition. However, the AD performance was improved when 50% of the recirculated liquid was replaced by fresh water. Another concern about using AD effluent as SS-AD inoculum is the transportation cost. Therefore, collocation of L-AD and SS-AD systems provides a solution to this issue [66]. Although reuse of AD effluent as SS-AD inoculum is beneficial for enhanced biogas production, the total volume and composition of the AD effluent does not decrease. Further management of the digestate from SS-AD is required to alleviate mass accumulation.

3.2 Dewatering

Many of the emerging technologies require either the liquid or solid fraction of AD effluent. The most common method to separate AD effluent components is dewatering, which has been shown to reduce AD effluent volume and improve solids handling [67].

Conventional methods for dewatering include centrifugation, screw press, and belt-filter press [57]. Typically, inorganic chemicals or organic polymers are added to improve separation via charge neutralization or molecular intercrossing [68]. Dewatering can provide an easier to handle solid fraction with much lower volume, and a liquid fraction mainly containing soluble nutrients [57]. After dewatering, the solid fraction of AD effluent contains concentrated levels of organic nitrogen and phosphorus, while the liquid fraction has more plant available TAN [5, 57]. This indicates that the solid fraction of AD effluent could be used as a slow release fertilizer, while the liquid fraction is more applicable for rapid nutrient supply during irrigation [5]. Prior to land application, AD effluent should be stored to reduce volatile methane and ammonia emissions [57]. Environmentally friendly land application techniques, such as trailing hose, trailing shoe, and direct injection should be used to mitigate TAN emissions and uneven nutrient distribution [57]. Although spray irrigation is a lower cost option for the liquid fraction, this method has a greater risk of TAN volatilization and may cause uneven nutrient distribution [57].

Feedstock variability influences water distribution within AD effluent, which affects the efficiency of dewatering methodologies. Water is distributed as free water (not attached to particles), interstitial water (water trapped within flocs), surface water (water held on the surface of solid particles), and bound water (intracellular water) [68]. For AD effluent with more interstitial water, dewatering

efficiency will likely be reduced. In fact, Lukehurst et al. [57] presented that conventional dewatering produced lower and more variable DM capture efficiency for anaerobically digested manure (13-68%) than the reported capture efficiencies for anaerobically digested sewage sludge (50-80%) [11]. This indicates that AD effluents may have compositional variability that reduces dewatering efficiency. Emerging dewatering methods include passive filtration, dissolved air flotation (DAF) and evaporation. For example, Xia et al. (2012) showed that a pebble filter could be used to remove 81-98% of TS from anaerobically digested dairy manure, producing a low solids filtrate (<2% TS). DAF is commonly used by treatment plants to concentrate anaerobically digested sewage sludge [11]. DAF has also been used to further reduce solids from AD effluent prior to membrane filtration (Section 4.3.3). Fuchs and Drosch (2013) indicated that 50% of the volume of the liquid fraction can be reduced by evaporation, which can lower transportation costs. Although these techniques are effective, filtration requires long residence times (>14 days), DAF requires high pressures (several atm), and evaporation requires high thermal energy demand (300-350 kWh/ton water removed)[69, 70]. Pilot scale studies are needed to compare the economic feasibility of emerging and conventional dewatering methods. Future work should also determine how AD operating parameters affect the water distribution within AD effluent, and use pilot scale studies to optimize dewatering processes. These studies are important because many emerging methods require the liquid fraction of AD effluent to have very low solids content.

4. Emerging techniques for management of the liquid fraction of AD effluent

The liquid fraction is 90-95% of the total mass of commercial AD effluents [25, 28, 30]. Therefore, the majority of emerging techniques are designed to handle the liquid fraction of AD effluent. After dewatering, the liquid fraction contains relatively low solids, significant amount of organics, nitrogen, and phosphorus. Most commonly, the water and nutrients in the liquid fraction are reused for irrigation and fertilizer [57]. However, there is debate about the environmental sustainability of spray irrigation of AD effluent (Section 2.3). To improve treatment of the liquid fraction of AD effluent, technologies have developed in recent years focusing on organics removal, water reuse, nutrient removal, recovery, and reuse.

4.1. Microbial Fuel Cells for organics removal

Microbial Fuel Cells (MFCs) are bioelectrochemical systems that convert organic material into electricity through exploitation of microbial metabolism [71]. MFCs consist of an anaerobic anode and aerobic cathode separated by an ion-selective membrane. In the anaerobic chamber, microbes oxidize organic matter, producing electrons and protons. Electrons are transferred through an external circuit, which can be used directly for electricity. The protons pass through the ion selective membrane, combining with oxygen (O_2) at the cathode to form water [71]. The most common method to treat organics in wastewater is the activated sludge process, where aerobic bacteria convert BOD/COD into microbial biomass and carbon dioxide [11]. While the activated sludge process is extremely effective and robust, it is energy intensive and produces large quantities of sludge [11]. Contrarily, MFCs produce little sludge, and also produce renewable electricity.

However, the success of MFCs may be limited by the biodegradability of organic matter within the AD effluent [71]. Because AD already stabilizes organic matter, there may be limited amounts of soluble organics available for MFCs. Although the liquid fraction of piggery waste AD effluent had COD levels as high as 4.1 g COD L^{-1} [72], it may not have enough readily biodegradable COD for successful organics removal and electricity production. For example, Ge et al. [73] showed that MFCs had a wide range of COD removal ($36.2 \pm 24.4\%$) and volatile suspended solids reduction ($46.1 \pm 19.2\%$) from digested sewage sludge, with a maximum produced power density of 3.2 W m^{-3} [73]. Contrarily, Inglesby and Fisher [74] achieved a much higher COD removal efficiency (63%) and a columbic efficiency of 5.2% when MFCs were loaded with effluent from the AD of *Arthrospira maxima*, a cyanobacteria. The success of this MFC was attributed to the presence of soluble organics, such as VFAs, which were not previously converted into methane during the AD process [74]. Based on the above studies, the composition of readily biodegradable organics affects the efficiency of MFCs for organics removal and electricity production. Furthermore, MFCs produce very little power, and scale up is difficult [71]. Therefore, future research should continue to evaluate MFCs for AD effluent treatment using scaled up modular

arrangements. Another method for organics removal is the membrane bioreactor (MBR). MBRs offer the advantage of filtering the biomass produced by the activated sludge process in one reactor, producing an effluent with better water quality [75]. The downside to this method is that membranes are easily fouled by suspended solids. However, development of MBRs for AD effluent treatment has been limited. Therefore, economic and life cycle analyses that compare MFCs, activated sludge technology, and MBRs are warranted.

4.2 Freshwater recycle

Freshwater is a limited resource that should be reused wherever possible. The liquid fraction of AD effluent could be used for irrigation in areas of drought, for animal drinking water, and as a form of process water in several industries. For example, AD has been frequently used as a low cost method to treat distillage wastes from bioethanol plants. Moreover, due to high moisture content, the liquid fraction of AD effluent could be recycled as the water source for ethanol fermentation. Zhang et al. [17] developed an integrated ethanol fermentation and biogas production process in which distillage wastes from the cassava-to-ethanol process were subjected to AD, and the liquid fraction of AD effluent was reused as a water source for the ethanol fermentation process. The advantage of this technique is that water consumption during bioethanol production could be reduced. However, it would require removal of inhibitors, including organic acids, sulfide, and ammonia, that can accumulate in AD effluent and could inhibit ethanol fermentation [17, 76, 77]. For example, when Zhang et al. [17] applied untreated AD effluent as the water source for fermentation, ethanol yield was 0.4% (v/v) due to the presence of acetic acid (0.84 g L^{-1}) and propionic acid (3.29 g L^{-1}). When organic acids were extracted from AD effluent, ethanol yield was 10.1% (v/v), similar to ethanol fermentation using tap water (10.3%) [17]. One strategy to reduce inhibitors is to use two digesters in series for enhanced removal of organic acids prior to recycling the liquid fraction of AD effluent [17]. Another issue in reusing AD effluent for ethanol production is that additional acid is required to decrease the pH from 8.0 to about 6.0. In order to address the issue, Wang et al. [78] proposed mixing the AD effluent with a portion of the stillage, which has a pH of around 4.2-4.6. Interestingly, Gao and Li [79] reported no inhibition while using the liquid fraction of

thermophilic AD effluent for ethanol production. Additionally, when Gao and Li [79] used AD effluent as a water source, ethanol concentration was 79.6 g L^{-1} , an 18% increase compared to that using freshwater. Moreover, similar integrated processes were reported for citric acid [80] or polyhydroxyalkanoates (PHAs) production [81].

One major issue not mentioned in the previous research is that sterilization of AD effluent is required to eliminate bacterial contamination prior to ethanol fermentation. This would significantly increase energy requirements and operational costs for an integrated AD-ethanol biorefinery. So far, all fermentation studies using AD effluent have been conducted at laboratory scale. Future studies should be conducted at large scale, and should assess economical methods for AD effluent sterilization. Furthermore, economic feasibility analyses should be conducted, especially at large scale.

4.3 Nutrient removal, recovery, and reuse

4.3.1 ANAMMOX

Nitrogen and phosphorus discharged to waterways may lead to fish toxicity, eutrophication, and hypoxia [82]. Therefore, nutrient removal is critical for effective AD effluent management. Anaerobic ammonium oxidation (ANAMMOX) is an engineered process that exploits slow growing microorganisms that convert ammonium (NH_4^+) directly to nitrogen gas (N_2) using nitrite (NO_2^-) as an electron acceptor and inorganic carbon as an electron donor [83]. The process does not require an external organic carbon source, drastically reducing costs compared to traditional nitrification and denitrification methods [84]. A typical engineered system is designed to first partially oxidize NH_4^+ into NO_2^- in a partial nitritation reactor (PNR) prior to exposure to ANAMMOX bacteria [85] (Figure 2).

Figure 2: Schematic of PNR-ANAMMOX for nitrogen removal from liquid fraction of AD effluent

In Figure 2, the liquid fraction of AD effluent is diluted into a PNR such that the NO_2^- concentration is suitable for ANAMMOX reactors. AD effluent can be introduced into ANAMMOX reactors to adjust the $\text{NH}_4^+/\text{NO}_2^-$ ratio directly. Finally, recirculation could be used to maintain optimal hydraulic retention times (HRTs). Several design strategies are employed for PNR-ANAMMOX, including the continuously stirred tank reactor (CSTR), membrane bioreactor (MBR), sequencing batch reactor (SBR), immobilized bioreactor, single reactor system for high ammonia removal over nitrite (SHARON), and simultaneous nitrification and ANAMMOX in vertical flow recirculating biofilters [85, 86].

Magrí et al. [85] provided a thorough review of the primary factors affecting the efficiency of ANAMMOX treatment of AD effluent from manure digesters. To summarize, NO_2^- , $\text{NH}_4^+/\text{NH}_3$, inorganic carbon, organic carbon, pH, temperature, and dissolved oxygen are the most important parameters that affect process efficiency (Table 4). AD effluent contains high levels of inhibitors such as TAN, and dilution or long HRTs may be required to sustain microbial cultures within the two-stage PNR-ANAMMOX process [85]. However, ANAMMOX microbes may be less sensitive than microbes in traditional nitrification and denitrification processes. For example, Obaja et al. [82] reported a toxic limit of $500 \text{ mg L}^{-1} \text{ NH}_4^+-\text{N}$ for nitrifying and denitrifying bacteria. Still, the ANAMMOX process requires low levels of organic carbon (Table 4). Therefore, removal of organics from the liquid fraction of AD effluent

may be required prior to PNR-ANAMMOX treatment [85]. Pilot scale PNR-ANAMMOX systems have removed high levels of nitrogen ($500 \text{ g nitrogen m}^{-3} \text{d}^{-1}$) from the liquid fraction of AD effluent [87].

Table 4: Limits for effective nitrogen removal by ANAMMOX (adapted from Magrí et al. [85])

Parameter	Concentration range (g m^{-3})
Ammonium (NH_4^+)	600-5900 ^{c,e}
Ammonia (NH_3)	2-150 ^{a,c}
Nitrite (NO_2^-)	100-182 ^a
Inorganic carbon	150-295
Organic carbon	142-242 ^a
Dissolved oxygen (DO)	1% ^f
pH	6.5-8.8 ^b
Temperature	35-40°C ^d
Phosphate (PO_4)	31-620
Sulfide (S^{2-})	1-64
Salinity	8800-30000

^areported levels of inhibitory concentration; ^boptimal at 8.0; ^cdependent on pH; ^dminimum temperature at 6°C; ^eloading rate in partial nitritation reactor (PNR); f.from Strous et al. [88]

Magrí et al. [85] assessed the feasibility of large scale PNR-ANAMMOX reactors for nitrogen removal from the liquid fraction of AD effluent from a manure digester. Their results indicated that the low organic C/N ratio and significant NH_4^+ content in the AD effluent made the process feasible. However, the authors also stated that elevated levels of sulfides and phosphates could negatively inhibit the process [85]. Despite these limitations, the PNR-ANAMMOX process had a lower specific energy requirement (MJ m^{-3}) than air ammonia stripping/ $(\text{NH}_4)_2\text{SO}_4$ absorption (Section 4.3.2), traditional biological nitrogen removal, concentration by reverse osmosis (Section 4.3.3), struvite crystallization (Section 4.3.4), steam ammonia stripping/ $(\text{NH}_4)_2\text{SO}_4$ absorption, and concentration by vacuum evaporation [85].

The compositional variation in AD effluent produced from different feedstocks (Table 1) will likely affect PNR-ANAMMOX effectiveness, and process optimization is necessary. If successful, application of PNR-ANAMMOX could provide harmless N_2 gas and an effluent with little nitrogen, while using lower energy than traditional nitrification and denitrification. However, dilution is critical to mitigate inhibitory limits of NH_3 , phosphates, and sulfides (Table 4). The effects of microbial contamination on ANAMMOX effectiveness during AD effluent treatment has not been evaluated. Furthermore, ANAMMOX may lead to a net loss of reactive nitrogen, leading to increased demands for

inorganic fertilizers. Life cycle analyses that evaluate the impacts of ANAMMOX treatment on the nitrogen cycle are needed.

4.3.2 Ammonia stripping

Biological methods for nitrogen removal are easily inhibited by compounds such as TAN, which is prevalent in AD effluent, and process kinetics can be severely inhibited by competing microorganisms. Also, biological processes do not recover the nitrogen in a usable form. Physiochemical methods can handle high levels of TAN, are not dependent on biological kinetics, and provide the added benefit of nitrogen recovery [89]. One of the most effective methods for physiochemical nitrogen removal and recovery is ammonia stripping and absorption.

Ammonia stripping is the phase change of NH_3 from the liquid to the gas, which is achieved when a liquid containing NH_3 comes in contact with air or steam that contains little or no NH_3 [11]. The process is affected by the air/liquid ratio, pressure, pH, and temperature, with improved efficiencies at high pH (10-11) and high temperature (70°C) [15, 89]. The liquid fraction of AD effluent contains high levels of TAN and relatively high pH (7-8), making it ideal for ammonia stripping [15]. Ammonia stripping has recently been evaluated for AD effluent produced from food waste [90, 91] and agricultural manure [15]. In one study, Guštin and Marinšek-Logar [15] applied a continuous ammonia stripping process to the liquid fraction of digested pig slurry. High pH was critical to the performance, because it increased the $\text{NH}_3/\text{NH}_4^+$ ratio of the AD effluent prior to the stripping processes. Removal efficiencies were as high as 92.8% of TAN and 88.3% of TN, and the authors concluded that continuous ammonia stripping could substantially reduce the area of land required for agronomic land application of AD effluent [15]. In a separate study, De la Rubia et al. [90] applied ammonia stripping to AD effluent from source-separated food waste, using synthetic biogas (65% CH_4 , 35% CO_2) instead of air as the gas phase. TAN removal (13-44%) was much lower than previous studies [15], likely because of low flow rates used during the process (0.25 and 0.375 $\text{L}_{\text{biogas}}/\text{L}_{\text{AD effluent}}/\text{min}$).

The NH_3 from the exiting gas phase can be converted to ammonium sulfate ($(\text{NH}_4)_2\text{SO}_4$) through dilute sulfuric acid (H_2SO_4) absorption [92]. In the absorption process, dilute H_2SO_4 reacts with NH_3

laden air to form $(\text{NH}_4)_2\text{SO}_4$ that can be processed and sold as a nitrogen fertilizer. Packed bed acid scrubbers use inert packing materials to provide surface for gas-liquid absorption. Packed bed scrubbers are known for high NH_3 removal efficiencies ($>90\%$) but exhibit high pressure drop. Acid spray scrubbers have lower pressure drop but have slightly lower NH_3 removal efficiency (71-81%). These technologies could be used for renewable $(\text{NH}_4)_2\text{SO}_4$ production from NH_3 laden air after ammonia stripping of AD effluent [93].

Downsides to ammonia stripping include decreased efficiency at low temperatures, high costs of heating, requirements for pH control and air/steam compression, foaming risk, and inability to remove NO_3^- or NO_2^- [92]. Additionally, residual solids in AD effluent could induce clogging in ammonia stripping towers [89]. Ammonia stripping is typically optimized to treat specific AD effluent, because AD effluent composition can vary depending on AD operating conditions and feedstock components [89].

Electrochemical resource recovery has been used as a pretreatment step prior to ammonia stripping/absorption of AD effluent [94]. Similar to an MFC, the electrochemical resource recovery system had an anode and cathode separated by an ion exchange membrane, but voltage is applied to induce current and ion exchange [94]. Desloover et al [94] applied high ionic strength on the cathode side through addition of cations (Na^+ , K^+ , Ca^{2+}). Voltage addition caused net flux of the non- NH_4^+ cations from the cathode to anode, followed by increased flux of NH_4^+ from the anode towards the cathode [94]. Using AD effluent, the authors demonstrated a nitrogen flux of $90 \text{ g nitrogen m}^{-2} \text{ d}^{-1}$ at a current density application of 20 A m^{-2} [94]. After the NH_4^+ flux to the cathode, alkaline conditions converted NH_4^+ to NH_3 and an ammonia stripping/absorption system achieved 83% removal efficiency [94]. Desloover et al [94] indicated significant energy was required (9 kWh kg^{-1} nitrogen) for air stripping and $(\text{NH}_4)_2\text{SO}_4$ production after electrochemical concentration of TAN.

Ammonia stripping requires significant energy for heating/compression, and requires chemicals for pH adjustment, making full-scale operation costly [92]. To reduce costs, process optimizations are being conducted. For example, Jiang et al [89] conducted a pilot scale study for ammonia stripping the liquid fraction of AD effluent from a dairy manure digester. They indicated that effective ammonia

stripping (70-80% removal rates) could be attained at low temperatures (35°C), and lime requirements for pH adjustment were not excessive [89]. Jiang et al [89] also stated that the process has the potential to reduce the amount of land required for dairy manure application. However, these claims should be validated at pilot scale at more AD facilities.

4.3.3 Reverse osmosis

Another physiochemical method for nitrogen removal is reverse osmosis (RO). RO membranes are used to remove and recover ions from solution [67], including NH_4^+ . RO systems have been utilized to retain NH_4^+ from fertilizer manufacture effluents [95], landfill leachate [96], tannery wastewater [97], mine wastewater [98], and ammonia scrubber effluents [99]. RO methods have also been applied for NH_4^+ removal from AD effluent (Figure 3). For example, Bilstad et al. [100] used RO for nitrogen recovery from digested manure. The process removed 95% of input total nitrogen, which was retained in the liquid fraction (concentrate), and was therefore considered a viable liquid fertilizer [100]. Additionally, RO is effective at high initial TAN concentrations, as demonstrated by Masse et al. [16] who achieved >99% NH_4^+ retention efficiency using AD effluent with high initial TAN levels (10,000 mg L⁻¹).

Figure 3: Reverse osmosis for removal of nitrogen from liquid fraction of AD effluent

The RO process is influenced by operating pressure, pH, and temperature. Higher pressures promote higher flux of AD effluent through membranes [67], while low pH produces higher proportions of NH_4^+ , which is retained in the concentrate [16]. However, AD effluent contains solids that can cause membrane fouling during RO [101]. Even after dewatering, the liquid fraction still contains suspended

solids that may clog RO membranes [101]. Therefore, sieving or dissolved air flotation (DAF) followed by membrane filtration (i.e. microfiltration, ultrafiltration) have been used to enhance RO efficiency [16, 70]. Also, acidification could be used to increase the $\text{NH}_4^+/\text{NH}_3$ ratio (Figure 3) [16]. However, pre-filtration and acidification may be costly, limiting the economic viability of RO. To combat fouling, Vaneeckhaute et al. [101] applied vibratory shear enhanced processing (VSEP) for macronutrient removal from the liquid fraction of AD effluent. Estimates showed that VSEP could reduce overall energy costs and the concentrate fertilizer was estimated to have a value of $\text{€}6.3 \pm 1.1 \text{ t}^{-1}$ fresh weight (\$8.6) [101]. Another method to improve the cost effectiveness of RO uses purification of byproduct streams in the concentrate. Aside from nitrogen, potassium and VFAs present in AD effluent are also retained by RO membranes [16]. Potassium (K) is an important fertilizer for many crops (Section 2.2). Because they are used in pharmaceutical, petrochemical, food, cosmetics, and tanning industries, retention of VFAs by RO could become a valuable practice [102], improving the economic viability of AD systems [104].

4.3.4 Struvite crystallization

Phosphorus is a valuable agricultural fertilizer. However, high purity phosphate mines are rapidly depleting [103]. These issues have prompted the reuse of AD effluent as an organic fertilizer for crop production (Table 2). However, land application of AD effluent with low N/P ratio may result in over application phosphorus and runoff (Section 2.3). To increase the N/P ratio, phosphorus could be removed and recovered by struvite crystallization. Struvite crystallization can help control the N/P ratio while simultaneously producing a slow release phosphate fertilizer [104].

Significant quantities of NH_4^+ and some soluble phosphorus (PO_4^{3-} , HPO_4^{2-} , H_2PO_4^- , H_3PO_4) are in the liquid fraction of AD effluent. Controlled nucleation (crystal birth) and crystallization (crystal growth) of struvite ($\text{MgNH}_4\text{PO}_4 \cdot 6\text{H}_2\text{O}$) occur under supersaturated conditions by the equimolar reaction below [105]:

1 Controlled struvite formation from digested wastewater treatment sludge has been widely
2 reviewed, and is affected by thermodynamic and mass transfer properties, pH, supersaturation, mixing,
3 crystal size, impurities, and seed addition [12]. AD effluent does not have significant quantities of
4 dissolved magnesium (Table 1); therefore, external sources, including magnesium chloride (MgCl_2),
5 magnesium oxide (MgO), and magnesium hydroxide ($\text{Mg}(\text{OH})_2$), may be necessary to induce struvite
6 crystallization. One example of a struvite reactor is the fluidized bed design in Koch et al.[106]’s patented
7 process, adapted and outlined in Figure 4. Small struvite particles, formed at the top of the reactor, slowly
8 settled into sections of reducing diameter as they aggregated [107]. The settling zone is where largest
9 struvite crystals were harvested. Use of this technique produced high-purity (99.8%) struvite crystals that
10 progressed from loosely aggregated to solidly bound pellets [107].

Figure 4: Staged fluidized bed reactor for struvite crystallization (adapted from Koch et al. [106])

Struvite crystallization of AD effluent from agricultural waste digesters has been difficult. This is because the dissolved phosphate content in digested agricultural feedstocks may also be too low for proper struvite crystallization. Therefore, external phosphate may be needed, which increases costs. In fact, Zeng and Li [108] showed that the $\text{Mg}/\text{PO}_4^{3-}$ ratio needed to be five times higher than the stoichiometric requirements to induce struvite crystallization from digested cattle manure. AD effluent also has high levels of calcium, especially when cow manure is the AD feedstock [109]. The calcium competes with magnesium to form calcium phosphate precipitates, which may not be as valuable for downstream processing, because they are amorphous rather than crystalline [109].

The issues associated with struvite production from agricultural AD effluents could potentially be overcome. Dissolved phosphorus content can be increased through chemical pretreatment. Yilmazel and Demirer [110] showed that acidification of the solid fraction of AD effluent produced a phosphorus enriched solution. Yilmazel and Demirer [110] attained nearly complete phosphorus (99.6%-100%) and partial $\text{NH}_4\text{-N}$ (3.3-65.6%) recoveries from the phosphorus enriched solution in an optimized fluidized-bed reactor [110]. To release phosphorus from calcium phosphates in struvite reactors, Zhang et al. [111] added EDTA as a chelating agent. Their results showed that EDTA successfully released phosphorus, but warned that higher EDTA levels contributed to breakup of struvite [111].

Based on these studies, nutrients from the liquid fraction of AD effluent could be recovered by struvite crystallization. However, the high cost of magnesium inputs is difficult to overcome. For example, it has been estimated that struvite is more expensive than conventional rock phosphate because of the high magnesium cost [112]. To combat this issue, Lahav et al. [113] applied nanofiltration to seawater and used the filtrate as the magnesium source for struvite crystallization. Struvite recovery remained high (>95%) despite the high content of calcium in most seawater salts [113]. A separate recovery technique is hydroxyapatite ($\text{Ca}_{10}(\text{PO}_4)_6(\text{OH})_2$) crystallization, in which dissolved phosphorus adsorbs to cheap seeding materials and crystallizes into hydroxyapatite [114]. The composition of hydroxyapatite is similar to phosphate rocks, indicating it could be integrated into existing fertilizer manufacturing [114]. The use of acidification as pretreatment [110] and low cost magnesium inputs [113]

could improve the cost-effectiveness of struvite crystallization. Future investigations should evaluate struvite crystallization from AD effluent at larger scale and conduct economic analyses for feasibility assessment for both centralized and farm scale AD systems.

4.3.5 Hydroponics

Irrigation of the liquid fraction of AD effluent is effective because it is simple and produces valuable crops. However, environmental and economic concerns may limit this practice (Section 2.3). Use of the liquid fraction of AD effluent for controlled cultivation of crops in hydroponic systems could ameliorate issues related to soil quality, because plants are grown in nutrient solutions without soil. Additionally, hydroponic systems are isolated, and unused nutrients are more easily controlled and can be recycled for reuse. In one study, Liedl et al. [115] showed that hydroponic production of lettuce was comparable using the liquid fraction of AD effluent from a poultry waste digester or commercial fertilizers. However, some fruits were less amenable to AD effluent, due to the low NO_3^- content. Liedl et al. [115] showed that the NH_4^+ in AD effluent needed to be converted into NO_3^- for successful hydroponic tomato production [115]. Furthermore, dilution of the liquid fraction of AD effluent may be required to alleviate NH_4^+ toxicity. Krishnasamy et al. [19] evaluated dilutions of AD effluent for the best growth of silverbeet in a hydroponic system. They found 20% AD effluent dilution mitigated NH_4^+ toxicity and provided the best conditions for silverbeet growth. However, silverbeet growth was higher in a commercial hydroponic growth solution [19]. There are few studies on the hydroponic production of plants using AD effluent as the nutrient solution. Additionally, these studies did not investigate the impacts of microbial contamination during hydroponic production, which are crucial to ensure fruits and vegetables are safe for human consumption (Section 2.3). Another option to integrate AD and hydroponics is to utilize CO_2 and waste heat produced during biogas combustion for hydroponic greenhouse operation.

4.3.6 Algae production

Similar to hydroponic plant production, algae growth is strongly dependent on light exposure, temperature, carbon dioxide (CO_2), and nutrients such as nitrogen and phosphorus. Unlike hydroponic

plants, algae biomass contains high lipids [116], which have potential for conversion to biodiesel [117]. The liquid fraction of AD effluent contains high levels of nitrogen and phosphorus (1000-3000 mg L⁻¹ TN, 400-500 mg L⁻¹ TP [118, 119]) that can be used to supply nutrients for growth of non-toxic, lipid-rich algae biomass. However, the turbidity in AD effluent may cause inadequate light penetration for sustained algae growth. For example, Wang et al. [120] demonstrated that the growth rate of *Chlorella sp.* in AD effluent declined linearly with increasing initial reactor turbidity. Furthermore, Park et al. [121] showed that *Scenedesmus sp.* was inhibited by NH₃ at low levels (100 mg L⁻¹). To reduce the effects of turbidity and inhibitory NH₃, most researchers have diluted the liquid fraction of AD effluent to initial TN levels of less than 200 mg L⁻¹ (Table 5). This practice also diluted turbidity for improved light penetration. While the growth rate of algae can also be improved by optimizing the nitrogen to phosphorus (N/P) ratio [122], low nutrient content promotes lipid production [118]. Therefore, studies have attempted to optimize the nutrient loading to maintain growth and high lipid yields. Figure 5 displays a theoretical process for algae production using the liquid fraction of AD effluent for nutrients, brackish water as the water source, and CO₂ supplemented from biogas combustion. This method could provide a cost effective method for AD effluent reuse.

Figure 5: Algae production from waste products of the AD process

Selected publications where AD effluent was used for algae nutrients are summarized in Table 5. The most highly studied strains for the use of AD effluent nutrients are the freshwater microalgae genera *Chlorella* and *Scenedesmus*. Strains of freshwater algae are known for high growth rates ($0.3\text{--}1.7\text{ d}^{-1}$) and high nutrient removal (60–95%) [123]. Some freshwater strains can also be used for biogas upgrading through CO_2 assimilation [124]. Furthermore, *Chlorella* and *Scenedesmus* may also be mixotrophic, consuming CO_2 via photoautotrophic metabolism while simultaneously consuming VFAs from AD effluent via heterotrophic metabolism [125, 126]. Despite the advantages of using freshwater strains, substantial freshwater requirements may limit process economics and sustainability [127]. To combat these issues, marine strains could be cultivated in AD effluent diluted with seawater or brackish water. Under an optimal dilution (6% v/v) of the liquid fraction of AD effluent in artificial seawater, biomass productivity of *Nannochloropsis salina* was $92\text{ mg dry weight (DW) L}^{-1}\text{ d}^{-1}$ with nearly 30% lipids, higher than most freshwater strains cultivated in AD effluent. When AD effluent was used for algae nutrients in scaled up open channel raceway cultivation systems, growth rates were lower compared to laboratory cultures. Reduced growth rates were attributed to restricted light penetration and culture instability caused by predators [119, 128, 129].

Table 5: Algae biomass production using the liquid fraction of AD effluent as nutrient source

Algae strain	Working Volume (L)	AD feedstock	TN in algae reactor		TP in algae reactor		Biomass Production		Lipid Content (% dry weight)	Reference
			Initial Concentration (mg L ⁻¹)	Removal (%)	Initial Concentration (mg L ⁻¹)	Removal (%)	Growth rate (d ⁻¹)	Biomass Productivity (mg DW L ⁻¹ d ⁻¹)		
<i>Chlorella sorokiana</i> <i>Chlorella minutissima</i> <i>Scenedesmus bijuga</i> <i>Chlorella sp.</i>	0.06-0.2 0.1	Poultry litter Dairy manure	80-100 ^a 120-250 ^{a,b}	50-80 ^a 76-83	7-8 ^a 15-30 ^{a,b}	70-75 ^a 63-75	NR 0.28-0.41	71-75 NR	3.5-9.6 9.0-13.7 ^c	[130] [120]
<i>Scenedesmus sp. AMDD</i>	0.1	Vegetable, cow, swine, algae waste	23-28 ^a	23-100	8.5-22 ^a	13-99	0.85-1.66	NR	NR	[131]
<i>Chlorella sp. UMN271</i>	0.15	Swine manure	200-250 ^a	22-34	10-55 ^a	23-71	0.50-0.65 ^a	0.04-0.08 ^a	7.5-10.9 ^c	[125]
<i>Chlorella sp. & bacteria</i>	0.5	Wastewater sludge	225 ^a	89 ^a	120 ^a	17 ^a	NR	30 ^a	NR	[132]
<i>Scenedesmus sp.</i>	1	Livestock waste	120	40-55 ^a	NR	NR	0.04-0.09	46-57	NR	[121]
<i>Nannochloropsis salina</i>	1	Wastewater sludge	80-640	87-100	11.4-91.4	98-100	0.344-0.644	68-92	21-36	[118]
<i>Synechocystis sp.</i>	1	Wastewater sludge	80-640	71.2-100	11.4-91.4	99-100	NR	41.3-150.9	11.8-13.5	[133]
<i>Chlorella sp.</i> <i>Micractinium sp.</i>	2	Wastewater sludge	197	51 46	3.5	95 95	0.13 0.14	66 ^a 53 ^a	NR	[123]
<i>Chlorophyceae & cyanobacteria</i>	8	Wastewater sludge	45-80 ^a	48-92 ^a	5-11 ^a	98 ^a	NR	234	NR	[134]
<i>Chlorella sp.</i>	12	NR	481	78	51	73	NR	37-59	NR	[124]
<i>Chlorella sp.</i>	20 ^d	Dairy manure	40-200 ^e	72-98 ^e	NR	55-58 ^e	0.016 ^d	22.8 ^d	9.3-10.8 ^e	[135]
Mixed green algae culture	40	Dairy manure	16.3-30.5	96	1.8-2.6	98	NR	17.5	10-29	[128]
<i>Spirulina platensis</i>	100	Pig waste	28.2	80 ^a	17.6	30 ^a	NR	19.9 ^f	4.2	[136]

^aestimated from chart data; ^bTotal Kjeldahl Nitrogen (TKN); ^cTotal fatty acids (TFA); ^dresults from pilot-scale study; ^eresults from laboratory-scale study; ^fg m⁻² d⁻¹

Use of nutrients from the liquid fraction of AD effluent for non-toxic algae growth has potential to become an effective reuse option. However, several issues undermine the development of this technology. First, AD effluent contains residual solids that limit light penetration [120] and levels of NH_3 that may be inhibitory for algae growth [121]. Furthermore, introduction of predators from AD effluent could inhibit the growth of algae, especially in open raceway ponds. However, few studies have applied unsterilized AD effluent as a nutrient source for algae growth [118, 119]. In both of those studies, *N. salina* growth was not inhibited in lab scale cultures [118, 119]. Some potential solutions to combat predators are screening for algae strains that are robust to microbial contaminants, rapid detection/tracking of predators, and filtration of AD effluent prior to use [129, 137]. However, these solutions may increase costs, limiting the incentive to use algae lipids for low value biodiesel. Additionally, use of the whole algae biomass for biogas production via AD eliminates the expensive lipid extraction step, lowering operational costs [138]. To further reduce operational costs and improve sustainability, use of brackish or seawater is preferred. Compared to digested sewage sludge, AD effluent from agricultural wastes has lower levels of pathogens and heavy metals, indicating the algae produced could be used for more high value applications, such as animal feed.

5. Emerging techniques for reuse of the solid fraction of AD effluent

After dewatering, the solid fraction of AD effluent can be stabilized by traditional methods, including drying, heat treatment, thermophilic aerobic digestion, beta-ray or gamma-ray irradiation, and pasteurization, before landfilling [11]. Specifically, thermal drying has been frequently used to increase total solids content to 98%, making it easy to transport and use the solid fraction as fertilizer or fuel [139]. However, traditional handling methods may be costly and environmentally intensive [10]. In order to improve management economics, several technologies are in development for effective treatment and reuse of the solid fraction of AD effluent.

5.1 Composting

Although composting is an older technology, it has emerged as a leading treatment technique for the solid fraction of AD effluent. Due to high temperature and aerobic microbial activities, composting

has the potential to remove pathogens and stabilize organic matter, thereby improving soil amendment properties. Composting also reduces the volume of AD effluent, improving transport economics. Composting requires a relatively low moisture content (40-75%) compared to AD. To reach the desired moisture content for successful composting, dewatering of AD effluent is normally required. For example, Lin et al [140] had to centrifuge raw AD effluent to 30% DM prior to mixing with yard trimmings in order to reach the 35-55% DM content required for composting. Some of the studies discussed in this section have also employed raw AD effluent and a co-composting bulking agent to attain the desired moisture contents. Thus, composting may offer an additional advantage for AD water reuse.

The success of composting is determined by several key factors, such as composition of composting materials, temperature, aeration, water content, pH, and turning. The C/N ratio of digested materials is recommended to be 20–30 for composting [141]. AD effluent usually contains a high nitrogen concentration, supplied from the degradation of organic nitrogen (proteins, amino acids). AD effluent should be co-composted with carbon rich and relatively dry bulking agents to balance the C/N ratio and moisture content. Previous studies have employed yard trimmings, vine shoot prunings, and sawdust as the bulking agents. Additives such as almond shell powder may also be added (1% fresh mass basis) to further increase the C/N ratio [13].

Aeration and composting temperature are also closely related. Sufficient air is necessary to support microbial activity, which generates heat and destroys pathogens, while too much air may result in temperature decrease [11]. Currently, there are two strategies to control aeration and temperature for composting AD effluent at small scale facilities. One strategy is to set a ceiling temperature and adjust aeration to maintain that temperature. For example, Das [142] co-composted AD effluent and yard trimmings and set a ceiling temperature of 55°C. In that test, when the operating temperature was higher than 55°C, air was provided continuously at a rate of 730 m³ air m⁻³ reactor d⁻¹. When the temperature was lower than 55°C, the compost was aerated intermittently (equivalent to 18 m³ air m⁻³ reactor d⁻¹). An alternative strategy is to control the aeration but not temperature. Franke-Whittle et al. [143] continuously provided aeration at a rate of 24 m³ air m⁻³ reactor d⁻¹, and the compost temperature achieved a high value

(>70°C) within two days, followed by rapid decline to lower temperatures (25-35°C). Despite the rapid temperature decline, the authors indicated the final compost in the laboratory trial was hygienically acceptable [143]. Turning is also used to provide oxygen and to homogenize composting materials [144].

For composting of AD effluent with bulking agents, the overall moisture content is normally maintained above 40% to support microbial growth. The amount of AD effluent used depends on the water holding capacity of the bulking agent and the mixing ratio of effluent and bulking agent. Composting material pH is usually controlled to be above 7 but lower than 9 to promote favorable microbial growth conditions. Additives such as nitric acid and elemental sulfur have been used to reduce pH of mixtures [13, 142]. Composting of AD effluent usually took 1–2 months and can be shorter than the regular composting process, due to the abundance of microbial communities in the AD effluent. Table 6 summarizes the conditions for co-composting AD effluents and bulking agents.

Table 6: Summary of composting operating conditions

Inputs for composting	Ceiling Temperature (°C)	Aeration, (m ³ air m ⁻³ reactor d ⁻¹)	Moisture (%w.b.)	Initial pH	Operation time (day)	Turning times	Ref.
AD effluent of 0.05–0.20 L/kg, Yard trimming. Additive: HNO ₃	55	730 at >55°C 18 at <55°C	55.6	7.64–9.39	43–44	2	[142]
Green wastes (52.4%), AD effluent (26.2%) and compost (21.5%)	-	24	64	7.10	63	3	[143]
Solid fraction of AD effluent (80–100%) and vine shoot pruning (0–20%). Additive: elemental sulfur (0.2%) and almond shell powder (1%)	65	Adjustable based on the temperature	>40	8.34–8.46	95	2	[13]
70% effluent and 30% yard trimmings	-	Forced aeration	30.5	8.83	28	daily	[145]
Manure effluent and sawdust	70.1	Manual rotation	>40	8.19	56	-	[144]
Effluent and yard trimming	-	Manual rotation	-	-	45	-	[66]

The end product, namely compost, may be used as a fertilizer/soil amendment. The quality of compost produced from AD effluent can be evaluated by stability and maturity parameters as well as the contents of organic matter (OM), N, P, K, heavy metals, and pathogens. Most previous studies showed that AD effluent from manure digester-based composts can have adequate OM, N, P, and K for plant growth. Additionally, the composition was similar to those found in other composts, such as those from agro-industrial wastes and manures [13]. Stability and maturity of compost can be assessed by measuring

1 the germination index and oxygen uptake rate. Troy et al.[144] studied composting of digested manure
2 and yard trimmings and showed that increasing the bulking agent from 0% to 40% significantly improved
3 the germination index from 59% to 83%, indicating the composts were phytotoxin free. The oxygen
4 uptake rate was also significantly reduced during composting, indicating that the compost was stable at
5 the end of the process. The leachate produced during the composting process can be recycled or
6 depending on its composition, may be viable dilution water for AD [146].

7 Heavy metals in AD effluent are not removed during the composting process. In two studies,
8 there were measurable amounts of heavy metals (copper, zinc) in composts, but the concentrations did not
9 exceed regulatory limits in the U.S. or Spain [13, 142]. Also, pathogens (*Salmonella*) may survive if the
10 composting process is too short. Franke-Whittle et al. [143] showed that a 34-day composting process
11 was not able to completely eliminate pathogens, while 63 days of composting was successful. Another
12 concern of land application of composted AD effluent is the possibility of odors. To overcome these
13 challenges, a longer composting process and more aeration may be needed to induce further stabilization
14 and reduce odorous emissions. Additionally, composting can be conducted in a closed chamber, where
15 volatile emissions can be captured and eventually treated [11].

16 5.2 Reuse of AD fibers for fuels

17
18 AD effluent can be used as a solid fuel after drying and compressing into pellets, which can be
19 used for incineration, gasification, and pyrolysis. Kratzeisen et al. [147] co-digested maize silage with
20 different field crops and animal manure, and dried the AD effluent to 9.2-9.9% moisture and pressed it
21 into pellets. The AD effluent fuel pellets were an excellent solid fuel alternative, with calorific values
22 comparable to wood pellets. Additionally, the flue gas emissions from burning AD effluent pellets were
23 within the limits for biofuels according to the German Federal Emission Control Ordinance. Moreover,
24 most (>90%) of the produced energy was provided by waste heat. Gnanendra et al. [148] studied
25 gasification using AD effluent briquettes generated from the AD of several lignocellulosic biomasses,
26 including agricultural residues and industrial processing residues. The produced syngas had consistent

composition and quality acceptable for engine operation [148]. Also, the process maintained lower gaseous emissions compared with incineration.

Pyrolysis is an effective method for the conversion of digested sludge into bio-oil, pyrolysis gas, and biochar [149]. The bio-oil can be upgraded into a crude oil replacement, the syngas can be used for energy, and the biochar is an effective soil conditioner. Cao and Pawlowski [149] performed an extensive review of sewage sludge pyrolysis and showed that syngas, bio-oil, and biochar formation is dependent on feedstock organic content (VS), temperature (400-1000°C), reaction time (0-45 min), and heating rate. In a case study, Cao and Pawlowski [149] indicated that AD plus pyrolysis of anaerobically digested sewage sludge was more energy efficient (84.7%) than direct pyrolysis of undigested sewage sludge (67.6%). Similarly, Li et al. [150] reported on pyrolysis of AD effluent after co-digestion of chicken manure and corn stover, and suggested an 85% increase of energy recovery efficiency. However, several issues remain to be addressed by further investigations. For example, the ash content of AD effluent was usually high (15-20%), which may cause disposal concerns [147, 148]. Additionally, AD effluent must be dried to a low moisture content (<10%) prior to gasification, pyrolysis, or direct incineration in order to maintain energy efficiency. Besides, these systems should be built on-site of or close to the AD facilities due to the production of odor [147].

AD effluent, especially that from dairy manure and lignocellulosic biomass, are rich in recalcitrant fibers. These fibers can produce glucose comparable to other lignocellulosic biomass using proper pretreatment methods, which makes it a potential feedstock for bioethanol production. Following alkali pretreatment and enzymatic hydrolysis of digested dairy manure, Teater et al. [151] achieved an ethanol yield of 14.7 g L⁻¹, a comparable value to switchgrass (16.6 g L⁻¹) and corn stover (18.1 g L⁻¹). Yue et al. [22] reported a 90% glucose conversion efficiency during enzymatic hydrolysis of alkaline pretreated AD fiber, and a 72% ethanol yield from the hydrolysate (based on theoretical ethanol yield). Correspondingly, the authors estimated that an integrated process of AD and cellulosic ethanol production could produce more than 1.67 billion gallons of ethanol from 63 million dry tons of AD fiber, generated from 120 million dry tons of cattle manure available annually in the U.S. Yue et al. [152] further showed

that dilute alkaline pretreatment was better than other pretreatment methods for treating AD fibers. The authors also showed that an continuously stirred tank digesters produced better AD fibers for ethanol production than plug flow digesters, because the process had 21% higher ethanol production and 33% higher net energy output. MacLellan et al.[153] and Yue et al.[154] investigated co-digestion of corn stover with swine manure and dairy manure, and integrated the AD process with ethanol production using the AD fibers. Net energy production was estimated in the range of 5.5-9.6 MJ kg⁻¹ dry raw feed. Although AD effluent glucose yields were acceptable, recovery and reuse of the pretreatment chemicals are required to make this process economically feasible [155].

5.3 Reuse of AD fibers for materials

Another interesting alternative is the utilization of the solid fraction of AD effluent for particleboard production. While grass-based particleboard is a promising renewable product, a hydrophobic layer on the surface of grass made up of wax, inorganic silicon, and fat, requires pretreatment [20]. Pretreatment is critical when using low-cost water-based adhesives such as urea, formaldehyde and protein/starch-based adhesives in particleboard production. AD could be used to pretreat grasses, since extractives are degraded during AD process. Zheng et al. [20] reported an integrated system producing biogas by AD of saline creeping wild ryegrass, and particleboard using the AD effluent. The mechanical and long-term 24-h water resistance properties (except for modulus of elasticity) of the particleboards were significantly improved compared to those from untreated grass [20].

In the U.S., many farmers have benefited from generating animal bedding materials through anaerobic digestion [156, 157]. The solid fraction of AD effluent is low cost and rich in fiber, making it an attractive bedding material. In fact, Alexander [157] reported that very little amounts of digested manure are sold in the U.S. because most is reused for animal bedding [157]. Digestate from SS-AD may be a better bedding material due to its lower moisture content compared with L-AD effluent. Despite the potential benefits, there are issues with the use of AD effluent for bedding. It was shown that AD effluents have considerable levels of bacteria, which could increase the threat of infection of cattle breast

tissue [21], which can increase milk contamination. Repeated changing of bedding materials is necessary to mitigate bacterial loads when using AD effluent fibers [21].

The production of cement and aggregate from waste sludge has been thoroughly investigated [158]. Direct use of dewatered sludge as an energy source in Portland cement kilns can completely oxidize organic pollutants in sludge, and immobilize heavy metals into the final cement [158]. Additionally, the requirements of natural resources such as clay and limestone are reduced, because sludge ash is utilized in the cement [159]. However, there is little research published on the use of the solid fraction of AD effluent as an energy amendment for the production of cement. In a life cycle analysis, digested sludge was compared with raw sludge for use as an energy supplement in cement manufacturing. Digested sludge had a lower heating value due to the lower organic content [160], which reduced its effectiveness for reuse in cement manufacturing. However, reuse of digested sludge for concrete manufacture was more environmentally sustainable than direct incineration of digested sludge [160]. These claims should be validated through a pilot scale study using the solid fraction of AD effluent.

6. Future recommendations

6.1 Emerging organic pollutants and treatment techniques

Besides pathogens and nutrients, growing attention has been focused on organic micro-pollutants in AD effluent and their harmful effects during land application. Studies have noted that some pharmaceuticals (e.g. Ibuprofen, Iopromide, and Diazepam), pesticides (e.g. m-Cresol), fragrances (e.g. Galaxolide), and detergents (e.g. Triclosan) are bio-transformable to some extent during the AD process, while others such as linear alkyl benzene sulphonates, nonylphenols, synthetic musks, estrogens (e.g. Estrone), and ampicillin are resistant to degradation in the AD process (Table 7) [161, 162]. Therefore, antibiotics used in dairy farming and detergents used in food could possibly exist in AD effluent. These may cause endocrine disruption in terrestrial or aquatic ecological receptors. Most of the studies in Table 7 focused on the elimination of organic micro-pollutants, while little work has been done to investigate their degradation pathways and fates (end products). Therefore, it is difficult to evaluate the risks of land

applying AD effluent that contains these pollutants. Another concern about organic micro-pollutants is that they may negatively affect AD efficiency. It was observed that some antibiotics such as ampicillin, florfenicol, sulfamethazine, and tylosin can inhibit AD microbes, retard degradation of other compounds, and decrease biogas productivity [163]. Understanding the effects of emerging organic micro-pollutants on the utilization of AD effluent is challenging. This is because there are technical hurdles in characterization of the vast number of pollutants during AD and land application of the effluent.

Advanced oxidation has been employed to reduce the concentration of emerging organic micro-pollutants in AD. Carballa et al. [164] examined the influence of ozone pre-treatment on sludge AD, and showed that emerging organic micro-pollutants were removed with efficiencies ranging from 20% to 99%. However, the dewatering properties of sludge were deteriorated when the ozone pre-treatment was applied. Other oxidative pretreatment methods, such as oxidation using air and Fe^{2+} at high temperatures and elevated pressures have been used for wet sludge disintegration [165], but may also have the potential to be applied for the removal of organic micro-pollutants in AD effluent. Incineration can be another alternative, but requires dewatering. One major concern of all the above mentioned methods is the cost. Therefore, feasibility studies are needed.

Table 7: Behavior of emerging organic micro-pollutants during AD and after land application

Pollutant	Formula	Application	Behavior during AD and after land application	Ref.
Ampicillin	$\text{C}_{16}\text{H}_{19}\text{N}_3\text{O}_4\text{S}$	Antibiotic	Not degradable, can be inhibitory to AD at high concentrations.	[163]
Bezafibrate	$\text{C}_{19}\text{H}_{20}\text{ClNO}_4$	Medicine	Moderately degraded in AD, unknown end-product	[166]
Bisphenol A	$\text{C}_{15}\text{H}_{16}\text{O}_2$	Resins	Moderately degraded in AD	[167]
Bis (2-Ethylhexyl) Phthalate	$\text{C}_{24}\text{H}_{38}\text{O}_4$	Plastics	Not tested in AD; possibly degradable in soil.	[168]
Butanone	$\text{C}_4\text{H}_8\text{O}$	Solvent	Not tested in AD; possibly volatile in soil.	[169]
m-Cresol	$\text{C}_7\text{H}_8\text{O}$	Pesticide	Moderately degraded in AD, unknown end-product; inhibited soil ammonia oxidizing bacteria.	[170]
Estrone	$\text{C}_{18}\text{H}_{22}\text{O}_2$	Hormone	Poorly degraded in AD, unknown end-product; persisting in soil.	[171]
Galaxolide	$\text{C}_{18}\text{H}_{26}\text{O}$	Fragrance	Moderately degraded in AD, unknown end-product; possible crop uptake.	[172, 173]
Ibuprofen	$\text{C}_{13}\text{H}_{18}\text{O}_2$	Medicine	Moderately degraded in AD, unknown end-product; no crops uptake.	[173, 174]
Naproxen	$\text{C}_{14}\text{H}_{14}\text{O}_3$	Medicine	Moderately degraded in AD	[167]

Toluene	C_7H_8	Solvent, fuel	Completely degraded in AD.	[175]
Triclosan	$C_{12}H_7Cl_3O_2$	Detergent	Moderately degraded in AD, unknown end-product; rapidly dissipated in soil.	[176, 177]

6.2 Pathogen regrowth

Although AD can be an effective method for pathogen stabilization, there is also evidence of insufficient pathogen destruction. For example, Alfa et al. [178] showed that while *E.coli* and *Shigella sp.* were effectively removed via AD, while species of *Salmonella* and *Klebsiella* survived. Additionally, Fu et al. [179] indicated that the AD process may only alter the culturable state of microorganisms, rather than killing them. The use of cattle manure AD effluent also poses risk, due to the wide use of antimicrobial drugs in cattle farming. This could result in formation of multi-drug resistant bacteria, as identified in AD effluent by Resende et al. [180]. Traditional processing methods may affect regrowth of harmful microbes from AD effluent. For example, regrowth of fecal coliforms, *E. coli* and *Enterococci* occurred after Chen et al. [181] dewatered thermophilic and mesophilic AD effluent by centrifugation [181]. In a similar study, Erkan and Sanin [182] showed that belt filter dewatering caused 0.6 log decreases in indicator organisms, while 0.7-1.4 log increases occurred after centrifugation. Contrarily, the introduction of effluent to the soil can improve bacterial decomposer communities and improve nutrient cycling [183]. Still, to reduce public health risks, development of low cost monitoring techniques via molecular (DNA, RNA) methods is critical. If the issue of pathogen regrowth is widespread, AD effluent may require further stabilization by energy intensive methods (thermophilic aerobic digestion, pasteurization). Still, the heating demands for further stabilization could be obtained via waste heat produced during the combustion of biogas. Researchers should continue to monitor the effects of pathogens, an issue that has limited the area available for land application (Section 2.3).

7. Summary and Conclusions

Widespread adoption of anaerobic digestion for bioenergy production may be limited by the massive quantities of AD effluent produced. Agronomic land application has been the primary AD effluent management technique. However, environmental restrictions due to pathogen accumulation, food safety, and nutrient runoff have necessitated the development of alternative technologies for AD effluent

1 handling. The economic and environmental concerns related to land application of AD effluent could be
2 combatted by using emerging treatment technologies. Aside from land application, there are few
3 techniques to directly reuse or manage AD effluent. While AD effluent is a viable inoculum source for
4 solid state anaerobic digestion, the solid-state anaerobic digestate is easy to handle and can be composted
5 for fertilizer.

6 Dewatering techniques such as centrifugation have been used at large scale facilities to partition
7 raw AD effluent into liquid and solid fractions that are easier to handle and transport. Recently,
8 evaporation has been used to further reduce AD effluent volume and DAF is an effective method to
9 remove solids. However, the high energy demands of mechanical separation and evaporation may limit
10 their adoption to large scale AD facilities that produce excess quantities of electricity and process heat.
11 Smaller scale facilities could use less energy intensive methods such as passive filtration [10, 69, 70].
12 Still, the variability of water distribution in AD effluent may negatively influence dewatering efficiency.
13 For instance, the colloidal particles in digested manure may not be easily separated without polymer
14 addition [184]. Therefore, techno-economic and life cycle comparisons of dewatering techniques are
15 needed.

16 The liquid fraction is commonly used for irrigation, but concerns about volatile TAN emissions
17 and nutrient runoff may reduce the acceptance of this activity. The water could be reused for industrial
18 processes such as bioethanol, but limited pilot scale research is available. Residual organics could be
19 converted to renewable electricity in MFCs. However, low quantities of degradable organics may limit
20 efficiency and power output. Thus, activated sludge technology seems most feasible for organics removal
21 prior to discharge. Dissolved nutrients in the liquid fraction provide the greatest opportunity for
22 valorization. Currently, physiochemical technologies such as ammonia stripping/absorption, reverse
23 osmosis, and struvite crystallization have successfully produced nitrogen and phosphate fertilizers from
24 the liquid fraction at pilot and full scale [90, 102, 108]. Digested manures have high TAN, indicating large
25 volumes of ammonium based fertilizers could be produced by ammonia stripping/absorption and reverse
26 osmosis. However, ammonia stripping/absorption has significant energy demands and reverse osmosis is

known for high capital costs. Thus, the success of these technologies will be dependent on the scale of operation and cost of energy. Struvite crystallization may not be feasible for digested manure and food waste because of low dissolved P content. Additionally, the need for exogenous magnesium may reduce economic feasibility. Reuse of nutrients for hydroponics and algae culture show good potential, but require significantly more research about food safety and microbial contamination, respectively. Additionally, limited pilot scale data indicate hydroponics and algae culture will be more applicable in the long-term. ANAMMOX appears to have potential for sustainable removal of N in the liquid fraction. However, the life cycle impacts of removing reactive N requires assessment.

Composting is an effective method to remove pathogens and reduce the volume of the solid fraction of AD effluent. Additionally, the production of compost offers considerable value. Furthermore, raw AD effluent can be integrated in the composting process and composting leachate can be used for AD dilution water. The major downside to composting is the cost of aeration and long treatment times. However, composting remains a feasible technology for management of the solid fraction of AD effluent. The solid fraction could also be used as a feedstock for fuels, but the cost of pretreatment (enzymatic hydrolysis, drying) may limit the economic viability of this practice. The use of AD effluent for animal bedding has increased in recent years. However, there is need to study the impact of bedding on animal health. Use of the solid fraction for particle board and cement production is intriguing, but there is a lack of pilot scale research. Table 8 provides a summary of the emerging methods for AD effluent treatment.

Table 8: Comparison of emerging technologies for treatment of AD effluent

Treatment Process	Adapted for	Scale	Advantages	Disadvantages
Reuse as AD inoculum	Raw effluent	Bench/Pilot/Full	Uses whole mass of AD effluent, stabilizes AD process	Total mass of AD effluent not reduced
Microbial fuel cells	Liquid fraction	Bench	Organics removal, electricity production	Low power output, limited pilot scale study
Freshwater recycle for ethanol	Liquid fraction	Bench	Reuse of freshwater, biofuel production	Microbial contamination,

production					limited pilot scale study
ANAMMOX	Liquid fraction	Pilot/Full	Produces harmless N ₂ gas, low energy requirements		Microbial contamination, Loss of reactive N
Ammonia stripping	Liquid fraction	Pilot/Full	High NH ₃ removal efficiencies, NH ₃ in outlet stream can be converted to fertilizer		High energy inputs, need for pH control
Reverse osmosis	Liquid fraction	Pilot/Full	High NH ₄ ⁺ removal efficiencies, NH ₄ ⁺ in retentate can be used as fertilizer, other products (VFAs, potassium) recovered in retentate		High membrane costs, high energy demands, need for pH control
Struvite crystallization	Liquid fraction	Pilot/Full	High NH ₄ ⁺ and dissolved P removal efficiencies, produces solid fertilizer		Need for Mg input, agricultural effluents have low dissolved Mg and P content
Hydroponics	Liquid fraction	Bench	Control over nutrient loading, production of valuable crops		Microbial contamination, food safety concerns
Algae production	Liquid fraction	Bench/pilot	Reuse of N, biofuel production		Microbial contamination, high costs of biomass processing
Composting	Raw effluent, Solid fraction	Pilot/Full	Pathogen removal, compost production		Long treatment times, air emissions
AD fibers for fuels	Solid fraction	Bench	Biofuel production		Need for pretreatment, high energy demands
Animal bedding	Solid fraction	Pilot/full	Complete reuse of fibers, low costs		Microbial contamination, animal health concerns
AD fibers for building materials	Solid fraction	Bench	Complete reuse of fibers, valuable materials		Need for pretreatment, high energy demands

Future work should attempt to identify the impacts of organic micro-pollutants and pathogens during and after land application of AD effluent. Overall, the combination of agronomic land application and emerging technologies could provide unique waste handling solutions to the growing bioenergy

industry. Further study at pilot and large scales could provide significant advances that promote AD adoption and environmental sustainability.

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