

Review

Gaseous Biofuels in Biorefineries: Biomethane

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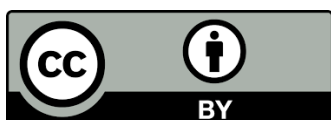
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Abstract

The global production of biomethane is rapidly emerging as a sustainable alternative to fossil fuels for energy generation. Biogas, where methane (CH₄) is usually one of the main components, is generated through the anaerobic digestion (AD) of various organic substrates, including animal manure, organic wastes, and wastewater. The efficiency of the AD process depends on key operational parameters and reactor designs that optimize microbial activity and gas yield. However, inhibitory compounds such as heavy metals, ammonia, and volatile fatty acids (VFAs) can significantly impact microbial metabolism and biogas production. To enhance substrate availability for microbial degradation, different pretreatment methods are often employed, as well as current advances in the AD focused on DIET enhancement and co-digestion technologies. This chapter provides a comprehensive overview of common substrates for AD, the impact of operational parameters, and strategies to mitigate the effects of inhibitory compounds. It also examines the latest advancements in reactor designs for



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effective monitoring and control of the digestion process. Additionally, the chapter explores biogas production for electrical energy generation, along with the purification technologies required to upgrade biogas to biomethane. Finally, it discusses the applications and value-added products derived from biomethane and its integration into circular economy frameworks. It involves the advantages and gaps hindering the full implementation of AD-derived energy worldwide.

Keywords

Anaerobic digestion; substrate pretreatment; biogas upgrading; circular economy

1. Introduction

In recent years, the global energy landscape has been subject to various changes due to the increase in energy demand, the depletion of fossil resources, and the crises generated by economic, political and social problems such as those faced by the COVID-19 pandemic and the war in Ukraine, since Russia, as the leading exporter of natural gas, has generated strong instability in global energy markets. In addition, with the impact of climate change, the need to switch to mostly sustainable energies has driven increased attention toward bioenergy derived from biomass [1, 2].

The energy sector is a major contributor to greenhouse gas (GHG) emissions, since it traditionally relies on fossil fuels. Currently, integrating biorefineries for energy generation has emerged as a key strategy to promote the efficient use of biological resources. In this context, AD technology plays a pivotal role, as it facilitates the production of biogas from a diverse array of feedstocks involving relatively low land use if based on residues, including crop residues, animal manure, the organic fraction of municipal solid waste, and sewage sludge [3, 4]. This approach can contribute to reducing greenhouse gas emissions by leveraging the potential of AD products, which would otherwise be released under natural waste decomposition into the atmosphere. However, the impact may be amplified under conditions requiring intensive transportation or inadequate management of the process [5].

The AD process involves the breakdown of organic matter through a complex series of biochemical steps: hydrolysis, acidogenesis, acetogenesis, and methanogenesis, each mediated by specialized microbial communities, including hydrolytic, acidogenic, acetogenic bacteria, and methanogenic archaea [6]. Maintaining a balance among these steps is crucial for the stable operation of AD systems [7]. The selection of reactor design and configuration is critical and must be based on the physicochemical properties of the substrates, desired operational parameters, and scalability of the process. Temperature, pH, organic loading rate (OLR), influent characteristics, and hydraulic retention time (HRT) influence biogas yields and process stability. Optimizing these parameters can enhance CH₄ production while reducing operational costs, thereby increasing the economic efficiency of AD systems in both industrial and field-scale applications.

Although CH₄ is the desired product, the presence of other compounds poses challenges for its application due to their potentially harmful and toxic effects. In addition, the diversity of microorganisms involved in the AD process makes its optimization difficult, and the variety and complexity of the substrates found in the waste may limit monodigestion, adding to the high costs

of upgrading and purification [8]. To address these challenges, various purification and upgrading technologies have been developed. For effective use of biogas, it must reach a CH₄ concentration of at least 90%; to qualify as biomethane, a concentration of 95% or higher is required [9, 10]. These upgrading technologies, which include water scrubbing, chemical and physical adsorption, biological processes, and cryogenic and membrane separation, can be categorized into *in-situ* and *ex-situ* techniques [11]. On the other hand, limited investment in some developing countries makes it challenging to adopt these systems and implement them on a large scale. Regulatory policies and the availability of raw materials are other key constraints hindering the development of this technology [12].

Biomethane holds significant promise due to its similarity to natural gas, enabling direct injection into gas grids for domestic use and electricity generation [13]. Furthermore, biomethane's potential applications extend to various types of vehicles, further underscoring its versatility and sustainability [14, 15], demonstrating a capacity to align with the distinct characteristics and objectives of multiple regions. In Europe, for instance, there has been an acceleration in the development of biomethane as part of initiatives such as the REPowerEU Plan. The primary objectives of this plan include reducing dependence on fossil fuels and ensuring energy security. The REPowerEU Plan aims to achieve growth in biomethane production, with the ultimate goal of completely replacing conventional natural gas and becoming independent of Russian supply by 2030 [16].

In the United States, the biogas bioconversion process is gaining traction as a viable alternative for the transportation and power generation sectors. This development is being driven by fiscal incentives and policies that support the circular economy [17]. Similarly, biomethane has excellent potential in Australia, as it can serve as a renewable energy source to replace natural gas, thereby encouraging sustainable waste management and promoting economic development in rural areas. The country's significant biomass resources have attracted substantial investment in the biomethane sector, contributing to energy security and environmental sustainability [11]. In countries like Mexico and Brazil, biomethane has enormous potential to reduce emissions and add value to organic waste, especially in the agricultural sector. Despite progress, the development of this sector still faces obstacles due to the lack of incentives, inadequate infrastructure, and the scarcity of regulatory frameworks. It is necessary to implement clear policies and increase subsidies to promote its use [18].

In today's global context, characterized by accelerated climate change and the looming scarcity of fossil fuels, investment in sustainable technologies for producing energy, fuels, industrial chemicals, and consumer products is increasingly crucial. The AD process in biorefineries represents a significant technological advancement, enabling the production of value-added products from organic waste in an economically viable and environmentally sustainable manner. By fully utilizing waste and valorizing the new waste generated during biomethane production and upgrading, this approach aligns with the circular economy model, which seeks to minimize environmental impact and promote the sustainable use of natural resources. Achieving the optimal development of a circular economy for biomethane and its derivatives will require the cooperation of all stakeholders, governments, institutions, producers, and consumers, to foster a paradigm shift towards the complete substitution of non-renewable energy sources and a more sustainable industry.

1.1 Anaerobic Digestion Process

As mentioned before, AD is a process where a series of bioreactions mediated by specific groups of microorganisms oxidize organic matter and convert it into biogas within a closed reactor. The steps in AD are sequentially linked, where the output of one pathway serves as the initial product for another (Figure 1) [4]. The process begins with the hydrolysis, in which the organic matter, primarily polysaccharides, lipids, and proteins present in the majority of AD feedstocks, is broken into soluble, low molecular weight compounds (fatty acids, monomeric sugars, amino acids) through the secretion of hydrolytic enzymes [19]. In the second step, acidogenic microorganisms convert the products of hydrolysis into alcohol and short-chain volatile fatty acids (VFAs) (propionate, acetate, and butyrate), H_2 , and CO_2 from the hydrolysis products [20]. Since acetate and H_2 are substrates for methanogens, these products can be metabolized directly by the methanogenic archaea to produce CH_4 finally. However, the products from acidogenesis, such as propionate and butyrate, are taken by acetogens to generate acetate and enter either the acetoclastic or hydrogenotrophic methanogenic pathways [6]. The performance of the reactor is highly influenced by the microbial communities [21], therefore the operational parameters must address the optimal performance of the different bacterial groups involved in the process to enhance optimal biogas production, including feedstock characteristics, OLR, temperature, HRT, C:N ratio, solid retention time, inoculum, and reactor design [4].

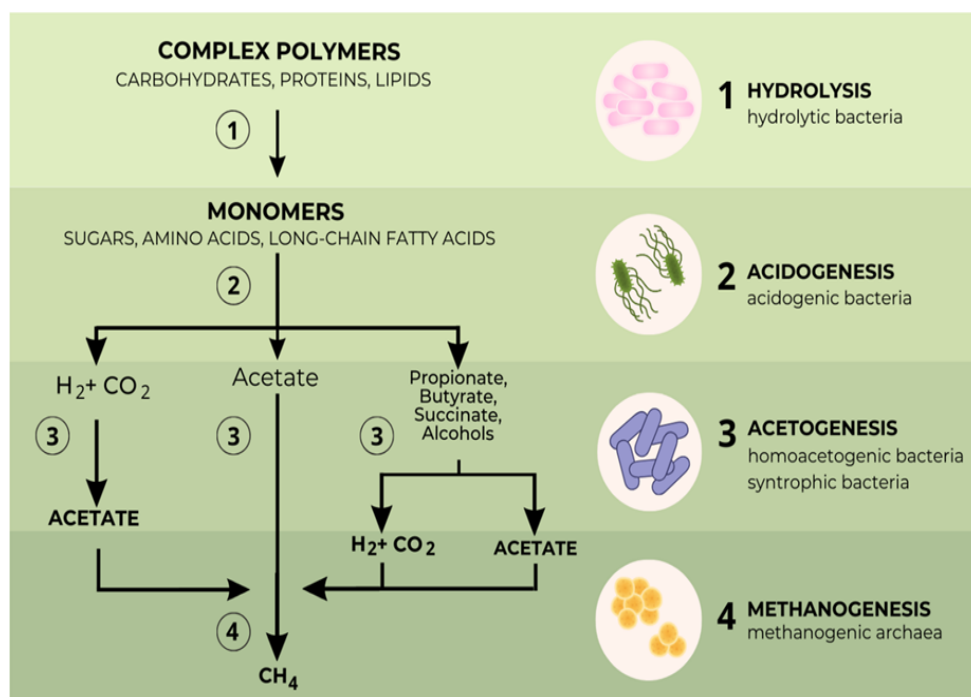


Figure 1 Anaerobic digestion steps, products and bacterial groups responsible for each step. 1) Hydrolytic bacteria break down complex polysaccharides into soluble, low-weight monomers. 2) Acidogenic bacteria generate acetate, VFAs, H_2 and CO_2 . 3) Acetogenic bacteria generate acetate, H_2 and CO_2 . 4) Methanogenic archaea take up acetate and/or H_2 and CO_2 , and generate CH_4 .

Microbial abundance throughout AD is typically dominated by Bacteria accounting for 93%, followed by a relatively small population of methanogenic Archaea 5.6% and eukaryotic organisms

1.1%, based on metagenomic sequencing. Main bacterial groups reported belong to Proteobacteria, Firmicutes, and Bacteroidetes phyla, with variations among Chloroflexi and Actinobacteria [22]. However, microbial communities remain uncertain, especially in large-scale fermentations, since most factors influencing microbial behavior remain unknown [23].

The hydrolysis stage is considered the rate-limiting step, dependent on substrate pH, particle size, production of enzymes, diffusion, and adsorption onto the substrate particles during the process. Hydrolytic microorganisms secrete a wide range of enzymes to degrade the polymers present [23], with both obligate and facultative anaerobes of the genera *Lactobacillus*, *Bacteroides*, *Propionibacterium*, *Megasphaera*, *Sphingomonas*, *Sporobacterium*, and *Bifidobacterium* [24].

During the acidogenic phase, Bacteroidetes, Chloroflexi, Cloacimonetes, Firmicutes, and Proteobacteria are the most common phyla. Particularly, *Firmicutes* species are relevant players in acetogenesis by utilizing simple and complex carbohydrates. Moreover, acetogenic bacteria include propionate decomposers such as *Syntrophobacter wolinii*, butyrate decomposers such as *Syntrophomonas wolfei*, and acid formers such as *Clostridium spp.*, *Lactobacillus*, *Peptococcus anereobius*, and *Actinomyces* [24].

In the methanogenesis step, acetoclastic archaea such as *Methanosaeta*, *Methanosarcina*, and *Methanothrix* are present, emphasizing the presence of *Methanosarcina* and *Methanothrix* as an indicator of the overall stability of the process. Additionally, *Methanosarcina* displays syntrophic relationships with acetate-oxidizing bacteria [22, 25].

1.2 Operational Parameters

The principal parameters controlled throughout AD depend on the number of stages, as various levels of complexity exist within the process. First, temperature is a critical factor, as it influences microbial growth rates, substrate utilization, physicochemical properties of the compounds, and overall stability of the process, affecting biogas production [4]. Digesters can be categorized by their operational temperatures and microbial communities as psychrophilic (10-20°C), mesophilic (35-39°C), and thermophilic (50-55°C). Psychrophilic conditions in AD exhibit low efficiency and are rarely applied [26], whereas mesophilic and thermophilic conditions are the most common operational modes applied at the industrial level, with acknowledged higher organic hydrolysis and conversion rates under thermophilic conditions [27-29].

The correct functioning of AD requires maintaining a balanced C:N ratio, which represents the quantitative correlation between carbon and nitrogen in the feedstock [4]. The optimal range for this ratio is typically between 20:1 to 30:1 [7]. While a single substrate may often have a lower C:N ratio, implementing additional substrates with complementary properties can help to balance this ratio [30]. In the AD process, carbon is an energy source, whilst nitrogen, primarily derived from proteins, is related to microbial growth. A low C:N ratio produces ammonia accumulation in the digester, leading to a pH increase. Conversely, a high C:N ratio can cause acidification, leading to a reduction of methanogenesis, destabilizing the process [4, 31].

A controlled pH range influences microbial growth in AD, especially for methanogens, which are highly sensitive to pH fluctuation. The pH is usually maintained between 6.5 and 7.2, depending on the substrate [4]. A low pH is often caused by the accumulation of VFA's, which are reported to inhibit methanogens. However, methanogens counteract through the production of CO₂, NH₃, and

HCO₃ [19]. Meanwhile, a high pH leads to the formation of free NH₃, which is toxic. Therefore, the stability of the process relies on the buffering capacity of the system [4].

The HRT corresponds to the average time, usually measured in days, that feedstock remains in the digester before being discharged. For the different operational temperatures, the average HRT has been reported as 10-14 days for mesophilic conditions, 10 days for thermophilic conditions, and up to twice that of thermophilic for psychrophilic conditions due to variations in VFAs accumulation. However longer HTR may be required, depending on the complexity of the substrate. The characteristics of the feedstock, such as particle size, total solids (TS), and volatile solids (VS) content, are critical to the AD process. The VS measures the biodegradable material in the feedstock that can be transformed into CH₄, while TS represents the total dry matter, both organic and inorganic [4, 32]. According to the German standard (VD1 4630), the substrate fed into the digester must be approximately 10 g VS/L, maintained at a range of 12-12 VS/L, with a TS of no more than 10% [33]. Moreover, control over the OLR sustains the equilibrium between the AD steps. It indicates the balance between the microbial community and the substrate, especially between acidification and methanogenesis [34]. Generally, thermophilic conditions allow a higher ORL than mesophilic conditions [35]. The main operational parameters considered when designing the system are the OLR and HRT, as well as the size and type of digester [4].

1.3 Inhibitory Compounds

A variety of compounds have been proven as inhibitors for the AD process, such as heavy metals (HMs), salts, aromatic compounds (AC), ammonia (NH₃) and VFAs, by shifting bacterial growth or reducing their activity, leading to reduced CH₄ yields [4]. Among the microbial community groups present in AD, methanogens are the most susceptible to inhibition. The inhibitors encountered throughout the process are introduced by the feedstock, as for HMs, salts and AC; or are generated via metabolic activities, including NH₃, and VFAs [32, 35].

1.3.1 Heavy Metals and Salts

Heavy metals, such as Cd, Cu, Cr, Ni, and Zn, are commonly present in agricultural and industrial wastewater, and tend to accumulate. While trace amounts of these metals are essential for enzymatic activity and as cofactors in AD, their effects can range from stimulatory to inhibitory depending on concentration (Table 1). Inhibition occurs when excessive HMs interfere with the spatial structure of key enzymes, such as hydrolytic and proteolytic enzymes, which are crucial for the AD process. The toxicity of HMs generally follows the order: Cu > Ni > Pb > Cr > Zn > Fe, with copper being the most toxic and iron the least. High concentrations of HMs can harm various microbial communities, but controlled additions of certain metals have enhanced CH₄ yields [36-38].

Table 1 Heavy metals present in AD, and their stimulatory and inhibitory concentration.

Heavy metal	Stimulatory concentration	Inhibitory concentration	Reference
Zn ²⁺	2 mg/L	50 and 100 mg/L	[37]
Pb	-	0.85 mg/L	[37]
Ni ²⁺	0.8 mg/L	>1 mg/L	[38, 39]
Fe ²⁺	205 mg/L	20,000 mg/L	[37]
Cu ²⁺	0-100 mg/L	>100 mg/L	[38]
Cd ²⁺	0.1 mg/L	>1.2 mg/L	[39]

Salt concentrations also have an inhibitory effect on AD, with methanogenic archaea being more vulnerable to salt stress than bacteria [40]. In a study conducted by [41], salt was gradually added to observe the responses of microbial groups and how it affected CH₄ production in AD, finding that NaCl at a concentration of 28.2 g/L was the limit before inhibition. In comparison, the optimal NaCl concentration reported was 2-4 g/L. High salt concentrations cause osmotic stress on microbial cell walls, resulting in desiccation or cell death; nonetheless, salt-tolerant microorganisms can be added to the system [41]. The presence of salts in AD substrates is mainly derived from table salt in food waste [40], where NaCl concentrations of 0.5~2 g/L were found to increase CH₄ yield from food waste, while 5 and 10 g/L NaCl reduced CH₄ yield by 36% and 41%, respectively [42].

1.3.2 Ammonia and VFAs

The AD systems commonly face the risk of NH₃ toxicity, especially when feed substrates are rich in nitrogen, and with a C:N ratio lower than 15, as in livestock manure, food waste, sludge, and microalgae. Ammoniacal nitrogen is produced by reducing organic nitrogen in the form of proteins and amino acids in AD. Although ammoniacal nitrogen is required for microbial growth, and the system buffering capacity, with concentrations between 50 and 200 mg/L are reported to have a beneficial effect on the AD process, concentrations of total ammonia nitrogen (TAN) above 1000-1500 mg TAN/L have been reported as a cause of AD failure. However, there is still no established inhibitory concentration for FAN, as diverse factors influence the microbial community resilience. Bacterial groups are reported to possess higher resistance to NH₃ toxicity than methanogenic archaea. The decline of methanogenic activity in AD performance often occurs at high TAN concentrations, which leads to the accumulation of VFAs, and further inhibition of methanogens, causing a decrease in pH and failure of the anaerobic process [43, 44]. Moreover, CH₄ production in AD systems feed with high-ammonia wastes is more prone to inhibition at thermophilic conditions, rather than mesophilic. Also, the increase in pH from 7 to 8 leads to an eightfold concentration of ammonia in mesophilic conditions, and more in thermophilic conditions [45]. Other inhibitory compounds besides ammonia that lead to pH decrease, are the VFAs, the most predominant intermediates during AD, and their accumulation indicates an imbalance between the sequential steps. At high ORL, the concentrations of VFAs tend to accumulate [46].

1.3.3 Other Toxic Compounds

Aromatic compounds (AC) are an underexplored group of inhibitors in AD, despite their ubiquitous nature. Their benzene ring structure makes their degradation under anaerobic

conditions thermodynamically challenging [47]. The primary sources of AC are lignin, present in lignocellulosic material, and aromatic amino acids in animal or plant proteins. The effect of various AC, such as phenyl acid, stems from toxicity and being stimulating for methanogens. For example, in reactors functioning under mesophilic and thermophilic conditions, aromatic substrates lead to phenyl propionic acid and phenyl butyric formation, which later hurt acetate degradation and methanogenesis [48]. Phenolic compounds alter the microbial cell wall permeability, and concentrations of >0.5 g/L cause AD failure; however, their presence in the feedstocks varies depending on the substrate utilized [49].

2. Common Substrates and Enhancing Methods

AD has been a leading method for processing sewage sludge and animal manure. Different types of biomass can serve as raw materials for biogas production. The composition of biogas and the amount of biomethane produced are directly affected by the type of feedstock used and the operational conditions of the digestion [50]. Due to their slow anaerobic decomposition, the high lignin content in organic substrates limits their use as feedstock for biomethane production. Traditional bioprocessing methods typically involve grinding lignocellulosic material before feeding it into an anaerobic digester, where polysaccharides and other complex compounds are converted into biogas [51]. Lignocellulosic agricultural biomass primarily consists of cellulose (35-50%), hemicellulose (15-25%), and lignin (10-15%), with the remaining components being oils, proteins, and ash. These lignocellulosic residues from agricultural sources can be used as feedstocks for producing advanced biofuels such as bioethanol, biohydrogen, biodiesel, and biomethane through both thermochemical and biochemical conversion processes [52].

Lignocellulosic materials are an underutilized source of fermentable sugars with significant potential for industrial applications. However, their decomposition is affected by several physical, structural, and compositional factors. To overcome these challenges and increase cellulose accessibility to microbial activity, pretreatment is necessary. A range of chemical, physical, physicochemical, and biological pretreatments are employed to break down these barriers to hydrolysis, resulting in several alterations to the lignocellulosic substrate, and an impact on the final CH₄ yield [53], as shown in Table 2.

Table 2 Principal lignocellulosic biomass in Biomethane production.

Lignocellulosic biomass	Operational conditions	Scale	Pretratement	Inoculum	Biomethane production	Geographical Distribution & Availability	Reference
Maize silage	21 days at pH 7.2, and 37°C	1 L lab-scale glass bioreactor	Microbial consortium with high cellulolytic activity	Cattle Slurry	393.365 dm ³ /kg of dry weight	North America, Europe (particularly Germany, Austria), parts of Asia; widely available as an energy crop with established supply chains	[54]
Corn stover	65 days at pH 7.5	1 L AD bottle	Ammonia in water at a fraction of 25% (v/v)	Sludge Inoculant	427.10 mL/g	North America, China, Brazil; abundant agricultural residue with seasonal availability following corn harvest	[55]
Oat straw	35°C for 150 days	2 L glass bottles digesters	=	Biodigestate from cow manure	0.320 m ³	Northern Europe, North America, Australia; available as an agricultural by-product with seasonal variations	[56]
Paddy Straw	45 days at 37°C, and pH 7	2 L glass biodigesters	NaOH-microwave treatment	Biodigested slurry	265-297 L/kg of dry matter	South and Southeast Asia (India, China, Thailand); abundant but seasonal, with challenges in collection logistics	[57]
Sugar beet silage	Inoculum to substrate (I/S)	2 L glass vessels, substrate	=	Digestate	350-400 m ³ /t of dry matter	Europe (particularly Germany, France), Russia, United States; available	[58]

	ratio of 0.5, at 35°C for 30 days					seasonally with established processing infrastructure	
Barley waste	S/I of VS 2:1, at 37°C for 90 days	Serum bottles	Samples were dried at 40°C for 48 h in an oven with forced air circulation and then milled	Cattle slurry	0.219-0.315 m ³ CH ₄ /kg of VS	Europe, North America, Australia; brewing industry by-product with consistent year-round availability in brewery regions	[59]
Raw straw (wheat, sorghum, and maize)	Continuously shaken at 50 r/min, 35°C and pH 7	1 L bottle	Crushed to a mean particle size of 1 ² mm by an electrical grinder	Anaerobic granular sludge with good biodegradability and methanogenesis ability	0.16 m ³ CH ₄ /kg of VS	Global distribution; wheat straw is common in Europe and North America, sorghum in Africa and Asia, maize straw worldwide	[60]
Wheat straw	35°C, and pH 7	1 L digester	=	Anaerobic granular sludge	0.16 m ³ CH ₄ /kg of VS	Europe, North America, Australia, parts of Asia; abundant agricultural residue with seasonal availability	[60]
Sugarcane waste	35 days at 38°C	Continuous stirred tank reactor	=	Biodigestate of AD	5-181 m ³ /t Fresh matter	Brazil, India, Thailand, Australia; highly available in sugar-producing regions with established collection systems	[61]
Forest residues	20 days at 55°C	118 ml serum glass bottles	Organosolv-pretreatment using ethanol, methanol, or	Inoculum from the municipal solid waste digester	0.23-0.34 m ³ CH ₄ /kg of VS	Scandinavia, North America, Russia; available year-round, with logistics	[62]

			acetic acid with sulfuric acid, or acetic acid as a catalyst			challenges in remote areas	
Cotton stalk cotton seed hull and cotton oil cake	37°C for 23 days	250 ml serum bottles	The paste cake and cotton stalks were pounded and ground	Anaerobic culture from the sludge digester	86 m ³ /t of waste	United States, China, India, Pakistan; seasonal availability following the cotton harvest Mediterranean countries, Brazil, United States (Florida); seasonal availability, with processing concentrated near juice production facilities Europe, North America, parts of Asia; wild- growing with potential for cultivation, currently limited commercial availability	[63]
Citrus waste	S/I VS ratio 1:2 at 37°C, for 82 days	118 ml serum glass bottles	=	Anaerobic sludge	0.137 m ³ CH ₄ /kg of VS		[64]
Nettle	35°C, S/I VS ratio of 1 for 130 days	2 L glass bottles digesters	=	Inoculum from cow manure	0.210 m ³ CH ₄ /kg of VS		[56]

Physical pretreatment methods are primarily employed to reduce the particle size of lignocellulosic biomass. Standard techniques include milling, grinding, and applying heat (through pyrolysis and other thermal pretreatments). Additionally, microwave irradiation and ultraviolet exposure disrupt the lignocellulosic structure, making cellulose and hemicellulose more accessible for further processing in AD [65]. Nonetheless, these methods encompass disadvantages, such as a significant economic barrier due to the generation of operational and maintenance costs, as well as the introduction of inhibitory compounds to the process [66]. Other emerging strategies to enhance biogas production involve co-digestion with different substrates and inoculation efficiency to increase AD performance [67], and the addition of direct interspecies electron transfer (DIET) promoting additives, such as activated carbon, biochar, and phenazine, which facilitate DIET, improving biogas yield [68].

2.1 Mechanical Pretreatment

Mechanical processing of lignocellulose is essential for improving conversion efficiency, particle compaction, and enzyme accessibility. By reducing the particle size of lignocellulosic material, the method enhances biomass conversion into biofuels without producing toxic subproducts [69]. This pretreatment increases surface area, improves bulk density and porosity, and optimizes the flow properties of the biomass. Techniques such as chipping, grinding, and milling are commonly used, with the degree of cellulose crystallinity reduction, polymerization, and surface area expansion depending on the milling method and biomass characteristics. Typically, raw material sizes range from 10-30 mm, while pulverizing can achieve sizes as small as 0.2-2 mm, depending on the nature of the biomass [70].

2.2 Pyrolysis Pretreatment

Pyrolysis is a thermal pretreatment process that involves heating lignocellulosic biomass to high temperatures. When the biomass is exposed to temperatures above 300°C, cellulose rapidly decomposes into char and gaseous products. As the temperature increases further, typically between 150-180°C, hemicellulose begins to solubilize, followed by the breakdown of lignin. In the context of AD, pyrolysis can enhance the digestibility of lignocellulosic material by altering its structure, making the remaining components more accessible for microbial conversion into biogas [56]. During the thermal process, a portion of the hemicellulose undergoes hydrolysis, leading to the formation of acids that are further broken down. The solubilization of lignin produces phenolic compounds, which can inhibit the activity of yeast, bacteria, and principally methanogens [71].

2.3 Irradiation Pretreatment

Irradiation initiates a chain-splitting process through the depolymerization of polymeric materials. Techniques such as electron beam, gamma ray, and microwave irradiation enhance the enzymatic degradation of lignocellulose. This treatment leads to the breakdown of polymer chains, the removal of lignin, and the disruption of the crystalline structure of cellulose. Using high-intensity electron beams on lignocellulosic substrates efficiently reduces the molecular weight and crystallinity of microcrystalline cellulose while increasing its surface area and accessibility [72]. Microwave irradiation offers an alternative to conventional heating methods. The intense and

focused heat generated by microwaves changes the orientation of polar molecule dipoles. This is a pretreatment method for materials like sludge, dairy manure, and kitchen waste. It is both efficient and easy to implement. When applied to biomass, microwave treatment modifies the cellulose structure, breaks down hemicellulose and lignin, and enhances enzymatic activity [73]. The irradiation process is a closed-system procedure that utilizes a gamma chamber, which limits the amount of substrate that can be processed at one time. Gamma rays transfer their energy to the atoms within the substrate, resulting in the formation of reactive radicals.

In addition to physical methods, chemical pretreatments, including solvents or enzymes, enhance biomass breakdown. These chemical methods are particularly effective at reducing the crystallinity of cellulose, which aids in the decomposition of biomass. However, these approaches are costly and may produce a mixture of compounds in a single product stream, making them less economical.

2.4 Acid Pretreatment

Acid hydrolysis can be carried out using dilute or concentrated acids, each affecting the lignocellulosic biomass differently [74]. This process primarily targets the hydrolysis of hemicellulose, especially xylans, breaking them down into simpler oligomers. Under acidic conditions, these oligomers are further hydrolyzed to produce monomers, along with byproducts such as furfural, and other volatile compounds.

During acid hydrolysis, solubilized lignin tends to precipitate quickly in acidic environments. When using strong acids for pretreatment, there is extensive solubilization of hemicellulose and more significant condensation of the solubilized lignin compared to dilute acid treatments [71]. This results in a more pronounced breakdown of hemicellulose and can lead to the formation of additional inhibitory compounds that may affect subsequent AD processes. The generation of volatile products during the solubilization of hemicellulose and the precipitation of solubilized lignin components are undesirable outcomes of strong acid pretreatment. These effects can reduce the overall digestibility of the biomass. Dilute acid pretreatment is considered a favorable technique because it minimizes secondary reactions and produces fewer degradation byproducts compared to stronger acid treatments. This helps to maintain higher biomass digestibility and reduces the formation of inhibitory compounds [75].

2.5 Alkaline Pretreatment

Alkaline pretreatment facilitates biomass solvation and saponification, leading to swelling that enhances its accessibility to enzymes and microbes. Strong alkali treatments can break down end groups, resulting in the formation of lower molecular weight compounds and the hydrolysis of dissolved polysaccharides, leading to a loss of these valuable components. Additionally, alkali extraction solubilizes lignin, causes its redistribution and condensation, and modifies cellulose crystallinity, which can reduce cellulose swelling and impede lignin removal [74, 76].

A crucial aspect of alkaline pretreatment is its ability to form a denser and more thermodynamically stable cellulose structure than native cellulose. For instance, xylan can be selectively removed using aqueous potassium hydroxide at low temperatures to prevent the peeling reaction [75]. Alkaline pretreatment is widely employed to enhance the enzymatic hydrolysis of lignocellulosic substrates, making it a key step in biomass processing.

Commonly used alkaline reagents include sodium, calcium, ammonium, and potassium hydroxides. Among these, sodium hydroxide (NaOH) is particularly valued for its exceptional delignification capability. It is crucial for achieving high substrate degradability, faster reaction rates, higher ethanol yields, and the absence of inhibitory effects. This makes NaOH a preferred choice in alkaline pretreatment processes to optimize biomass conversion efficiency [75].

2.6 Biological Pretreatments

Biological pretreatment involves using microorganisms, such as brown rot fungi, white-rot fungi, and soft rot fungi, to decompose lignocellulosic biomass [77]. White-rot fungi are particularly effective, as they break down both cellulose and lignin simultaneously. Some species, like *Ceriporiopsis subvermispora*, selectively degrade lignin in softwood and hardwood, making them valuable for targeted lignin removal. These fungi also show significant potential for bioremediation, capable of degrading persistent pollutants, including chlorinated and heterocyclic aromatic compounds, various dyes, and synthetic polymers [78]. Biological pretreatment is cost-effective, requires low energy input, and operates under mild physicochemical conditions. It also produces valuable by-products and generates fewer inhibitory substances, enhancing its suitability for sustainable biomass processing [79]. There are several techniques within biological pretreatment, as microaerobic pretreatments that use controlled oxygen levels to stimulate microbial activity; ensiling, which involves storing biomass in anaerobic conditions to promote natural fermentation; composting as a controlled aerobic process where organic waste is decomposed by microorganisms; digestion phase separation for different stages of AD to optimize microbial activity; microbial consortia that utilize mixed cultures of bacteria and fungi to achieve a more comprehensive breakdown of biomass; aerobic pretreatment with bacterial and fungal which employs specific microbial cultures under aerobic conditions to enhance lignin degradation and improve overall biomass digestibility [80]. These methods highlight the versatility and effectiveness of pretreatments in processing lignocellulosic biomass for AD.

Wastewater treatment plants are designed to process the municipal and industrial wastewater to minimize the environmental risks of releasing treated water into natural bodies. Using wastewater in AD can reduce the cost of wastewater treatment by generating renewable energy and minimizing waste disposal costs. Industries can also benefit financially by turning a waste stream into a resource [81]. Wastewaters are a valuable source of organic matter, which can be converted into biogas with a significant proportion of biomethane through AD. This not only recovers energy but also reduces the reliance on fossil fuels. AD provides an effective solution for treating wastewater, particularly those with a significant quantity of organic loads, by converting these wastes into biogas [82]. The use of AD to process wastewater reduces greenhouse gas emissions. It also helps control odors and reduce the pollution potential of wastewater. By integrating AD into wastewater treatment processes, communities can promote sustainable development by enhancing resource efficiency, reducing environmental impacts, and supporting renewable energy goals [52].

2.7 Co-Digestion Technologies

Co-digestion technologies in AD involve the simultaneous treatment of multiple organic substrates. These technologies optimize the microbial breakdown of various feedstocks, such as

livestock manure, food waste, agricultural residues, and industrial organic byproducts. Advanced approaches, such as two-phase AD, separate the hydrolysis and acidogenesis phase from the methanogenesis phase to better manage different feedstock characteristics and improve overall stability and biogas yield [83]. Co-digestion technologies also incorporate monitoring and control systems to manage key parameters like temperature, pH, and the C:N ratio of the substrate, ensuring optimal conditions for microbial activity. By combining various substrates, these technologies not only increase biogas yield up to 25-400% compared to mono-digestion but also enhance the economic viability and sustainability of waste management practices [67, 84].

Co-digestion offers several key advantages in AD, including improvement of process stability by adjusting the C: N ratio; enhanced buffering capacity, which helps maintain optimal pH levels; better flow and mixing properties of the digester contents; increased availability of essential nutrients, promoting microbial activity and the breakdown of substrates; reduced production and impact of inhibitory or toxic substances that can harm the microbes responsible for CH₄ production; improved quality of the digestate, making it more suitable for land application as a fertilizer; a synergistic interaction among diverse microbial communities, which boosts biogas yield; more stable and efficient digestion performance; and the ability to handle higher OLR [81, 85-87].

Most biogas plants that process agricultural waste explore co-digestion by combining cow, pig, and chicken manure with additional feedstocks like organic waste of farm industries, energy crops, food waste, and municipal biowaste. This strategy is designed to enrich the organic content of the input organic material, leading to a higher biomethane output [88].

For significant biomethane production, co-digestion accelerates the overall process kinetics, leading to faster decomposition of organic material. The balanced availability of nutrients, reduction and dilution of toxic compounds, appropriate moisture content, and robust buffering capacity all contribute to the effective and efficient operation of the AD process.

2.8 Inducement of Direct Interspecies Electron Transfer (DIET)

As mentioned, the AD process is divided into four sequential steps (Figure 1). Particularly, during the acidogenesis phase, short-chain fatty acids (SCFA) or VFAs are produced as precursors in the methanogenesis process. However, their accumulation can inhibit methanogenesis. The oxidation of SCFAs such as propionate, butyrate, and acetate is thermodynamically unfavorable ($\Delta G^\circ = 53$ kJ/mol). The energy required for these reactions is -15 kJ/mol, which is insufficient [89]. This energy limitation is mainly due to the dependence on hydrogen or formate as electron carriers, which is slow and requires a syntrophic partner to be effective. However, the oxidation of SCFA to methanogenic precursors (acetate, H₂, and formate) becomes thermodynamically favorable ($\Delta G^\circ = -20.4$ kJ/mol) when interspecies electron transfer (IET) mechanisms are involved. IET can occur in two ways: indirect interspecies electron transference (IIET), using metabolites such as hydrogen or formate as a transporter (use of proton as a terminal electron acceptor), and direct interspecies electron transference (DIET) [90]. In particular, DIET has emerged as a key process that enhances electron flow between syntrophic partners, enabling more efficient degradation of SCFAs and improved CH₄ production [91].

Whereas mediated interspecies electron transfer (MIET) depends on diffusible intermediates, DIET allows the direct transfer of electrons from one microorganism to another without needing diffusible intermediates such as hydrogen or formate [92]. In this process, syntrophic

microorganisms export electrons generated during respiration directly to partner species that serve as electron acceptors, facilitating more efficient energy sharing under anaerobic conditions [93].

This direct cell-to-cell interaction significantly accelerates electron transfer compared to MIET. DIET can occur through three main mechanisms: electrically conductive pili (e-pili), outer membrane c-type cytochromes, and conductive materials [94]. The use of conductive materials such as iron minerals, granular activated carbon, and biochar has recently been explored as a strategy to stimulate DIET and enhance methane production in anaerobic digestion systems [95]. For instance, [96] reported that adding magnetite to AD with high salinity organic waste as substrate has been found to increase the CH₄ production. Magnetite can absorb some salts, and with the addition of magnetite, there was a remarkable conversion of organic matter to CH₄; the production rates experienced boosts of 15.2% and 47.4%, respectively. The microbial diversity found was *Pseudomonas*, *Soehngenia*, *Thermanaerotherix daxensis*, *Dethiosulfatibacter*, *Halomonas*, *Desulfosporosinus*, *Desulfovibrio* and *Acetobacterium*.

Similarly, [97] evaluated the use of granulated activated carbon (GAC), which substantially reduced the lag phase and mitigated the accumulation of VFAs, a common bottleneck in overloaded digesters. In addition, the CH₄ production was higher compared with the control. The microbial diversity found in this study was *Firmicutes*, *Bacteroidetes*, *Synergistetes*, *Syntrophomonas*, and with the addition of GAC, a notable presence of fermentative bacteria was found: *Aminobacterium*, *Tepidimicrobium*, *T78*, *Pelomaculum*, *Ruminococcus*, *Clostridium*, and *Caldicoprobacter*, as well as the methanogens *Methanosarcina* and *Methanoculleus*. These microorganisms are commonly found in syntrophic associations, with *Methanosarcina* capable of participating in DIET when in contact with conductive surfaces.

Further supporting this, [95] reported that the use of food waste in AD, coupled with adding biochar made from wood waste created from pyrolysis at 750°C for 60 minutes, had a 21.5% increase in CH₄ production compared to the control. Demonstrating that biochar addition promotes the degradation of VFAs, contributing to improved methanogenic efficiency.

Despite its promise, several challenges hinder implementing DIET in AD. One major issue is verification and monitoring, as many studies attribute improved performance to DIET without directly confirming its occurrence [98]. Current detection methods include transcriptomics and proteomics, visualization techniques such as microscopy of nanowires, and electrochemical analyses like cyclic voltammetry. Additionally, the complexity of microbial interactions poses a challenge, since not all syntrophic microbes can DIET, and multiple electron transfer mechanisms, including DIET and MIET, may operate simultaneously [99]. Finally, the field lacks standardized protocols for effectively promoting and measuring DIET across different digester configurations and waste substrates, which hampers reproducibility and extensive scalability [100].

3. Reactor Types for Methane Production at Industrial and Field Scales

Selecting the appropriate reactor type must be tailored to the specific physicochemical properties of the substrate, desired operating parameters, and process scalability. To optimize AD for CH₄ production, it is crucial to understand the effect of operational conditions such as temperature, pH, OLR, influent characteristics, and HRT [101]. These parameters are critical for achieving optimal biogas yields and maintaining process stability. Choosing the right reactor type is fundamental for balancing trade-offs between biological optimization and economic sustainability

[102]. Overall, optimizing reactor performance requires a multi-faceted approach that combines biological, chemical, and technological interventions to balance CH₄ production efficiency with operational stability and economic viability.

3.1 Up Flow Anaerobic Sludge Blanket Reactor (UASB)

The UASB reactor is a widely used system for AD, characterized by the retention of microorganisms as granular sludge [103]. In this type of reactor, the influent is introduced at the bottom. It flows upward through the granular sludge layer, where the matter is degraded by the anaerobic microorganisms within the sludge. Granules in UASB reactors consist of dense aggregates of anaerobic organisms, creating a highly efficient environment for the degradation of complex organic matter [104]. The proximity of these microorganisms within the granules enables efficient substrate transfer and enhances metabolic cooperation. This results in higher CH₄ yields and more stable reactor operation under high organic loading conditions. The produced biogas rises and accumulates at the top of the reactor, where it is captured by a gas-liquid-solid separator [105], allowing the biogas to be extracted and separated from the liquid effluent and the sludge settled at the bottom of the bioreactor.

The UASB reactor offers multiple advantages, such as low sludge production and the potential to generate biogas with significant energy value [106]. Its design is especially suited to treating wastewater with high organic loads, supported by the dense microbial population within the granular sludge [107]. However, challenges such as the instability of the sludge granules can compromise reactor performance. Factors including influent composition, temperature, and flow rate can affect granule stability, and under high OLR and low HRT, solids accumulation and granule deterioration may occur, reducing CH₄ production and reactor stability [108].

To mitigate these issues, various modifications have been proposed. One notable approach is pre-acidification, which partially degrades organic matter before it enters the UASB reactor, thus reducing solids accumulation and improving the stability of the sludge granules [109]. This method is particularly beneficial under conditions of high organic load and short HRT, where granule deterioration is a critical concern.

To address challenges associated with high OLR, immobilized *Aspergillus sydowii* beads were utilized. Incorporating these beads significantly increased the reactor's tolerance to high OLRs, reaching up to 25.0 kg/(m³d) compared to 13.3 kg/(m³d) in the control reactor. This improvement resulted in a notable increase in COD removal and CH₄ production. The beneficial effects are attributed to the fungi's ability to inhibit the accumulation of VFAs and stabilize the pH [110].

More recent studies have employed artificial intelligence to enhance predictions of biogas production. Two AI-based models, a backpropagation artificial neural network (ANN) and a multilayer adaptive neuro-fuzzy inference system (ANFIS), were developed to estimate biogas production in a UASB reactor. These models used input variables such as influent chemical oxygen demand (COD), pH, effluent suspended solids, turbidity removal, oil and grease removal, COD removal, phenol removal, VFAs in the effluent, and alkalinity. The results demonstrated that both models provided accurate predictions of biogas production. By identifying and adjusting key variables affecting CH₄ production, these models can help maximize reactor efficiency, minimize CH₄ losses, and enhance overall system stability, leading to increased CH₄ production from treated wastewater [111].

[112] evaluated the performance of a full-scale UASB reactor from the Mudor Wastewater Treatment plant located in Accra, the capital city of Ghana, which comprises six UASB reactors operating in parallel with a capacity of 18,000 m³/day for municipal wastewater treatment in Ghana. Over 35 weeks, the reactor achieved 93% COD and 98% BOD removal, indicating effective organic matter degradation. Biogas yield was 0.2 m³/kg COD removed, with an average daily production of 831.6 ± 292.7 m³, 64.7 ± 11.9% of which was CH₄. Notably, 35% of the CH₄ remained dissolved in the effluent, pointing to opportunities for improved gas capture, with an overall capacity to serve approximately 100,000 inhabitants.

3.2 Continuous Stirred Tank Reactor (CSTR)

The CSTR reactor is a type of tank equipped with agitators that can operate continuously or intermittently. Effective mixing in a CSTR ensures homogeneity, optimizes contact between microorganisms and substrates, and maintains a constant temperature throughout the reactor [113]. This helps prevent issues such as microbial flocculation and the accumulation of VFAs, which could inhibit biogas production. Microbial flocculation can lead to the formation of dense microbial aggregates, making it difficult for substrates and nutrients to transfer to microorganisms, thus reducing their metabolic efficiency. Simultaneously, excessive accumulation of VFAs can lower the pH of the system, leading to acidification. This acidification inhibits the activity of methanogenic archaea, the microorganisms responsible for the final step in biogas production, resulting in decreased CH₄ yield and overall biogas production [114].

To enhance the performance of the CSTR, research has investigated the use of biochar as a mediator in the AD of cornstalk residues. The primary issues addressed were the high CO₂ proportion in the biogas, low gas production efficiency, and process instability due to the accumulation of VFAs and ammonia (NH₃-N). This resulted in a biogas production rate of 1.40 L/L/d and a 7.4% increase in CH₄ content, while CO₂ decreased by 5.9% compared to the system without biochar. Additionally, biochar facilitated the degradation of VFAs, reducing propionic acid concentration by 55.7%, and promoted greater microbial abundance, which was positively correlated with environmental parameters, thereby enhancing overall process efficiency [115].

In long-term AD studies of switchgrass (SG) in semi-continuous CSTRs, performance was enhanced by operating at an optimal OLR of 1.0 g VS/L/d. This resulted in a CH₄ yield of 157 mL CH₄/g VS, representing 35% of the theoretical yield and an energy recovery of 38%. This improvement was attributed to process stability at this OLR and adapted microbial communities effective in acetate oxidation [116].

The performance of the CSTR improved through anaerobic co-digestion of corn stover pretreated with urea and chicken manure, achieving volumetric methane production (VMP) rates of 2.160 and 1.616 L/L/d under OLR of 4.2 and 6.3 g VS/L/d, respectively. This approach increased VMP by 32.8%-89.6% and 27.8%-96.4% compared to other reactors, due to the synergistic effect of biochar and urea pretreatment, which enhanced the system's buffering capacity and the degradation of lignocellulosic biomass. Urea pretreatment breaks down complex lignocellulosic structures like cellulose and hemicellulose, making the compounds more accessible to microorganisms during AD. Biochar provides additional surface area for microbial attachment, facilitates electron transfer, and improves microbial activity [117].

[118] studied the effect of agitation time on a full-scale research biogas plant at “Unteren Lindenhof” of the Honenheim University containing two identical CSTR digesters with a volume of 923 m³ each. Considering a start-up phase of 25 days, mixing times from 10 to 5 and to 2 minutes, it yielded a gas production of 74 m³/h, 65 m³/h and 63 m³/h, having a reduction of 15%, which does not endanger biogas production by an uneven distribution and noticing that less mixing time resulted in 85% less energy consumption.

3.3 Internal Circulation Reactor (IC)

The IC reactor features a built-in circulation system, typically driven by agitation mechanisms or hydraulic flow, to ensure uniform mixing. This promotes efficient contact between microorganisms and substrates, enhancing the digestion process. In this setup, influent is introduced at the bottom, biogas accumulates at the top and is extracted through a piping system, while digestate is removed from the reactor's base. The internal circulation creates a homogeneous mixture, optimizing microbial activity [119]. IC reactors are widely used for organic waste management in wastewater treatment, agricultural waste digestion, and biogas production [120]. However, IC reactors present challenges, including requiring precise control of flow and internal mixing. They are also prone to solid accumulation, which can obstruct circulation [121].

A study focusing on the improvement of the anaerobic IC reactor aimed to optimize its hydraulic and biochemical performance by integrating accurate mathematical models of Incremental Size Continuous Reactors (ISC) and Anaerobic Digestion Model No. 1 (ADM1). The simulations accurately replicated experimental data, demonstrating that the model effectively predicts how the reactor handles variations in organic loading and maintains process stability. By accurately predicting the reactor's response to different loads and operational conditions, the process stability and overall efficiency of AD can be enhanced, maximizing CH₄ production [122].

In a pilot-scale study of the internal circulation anaerobic reactor, the use of anaerobic granular sludge as inoculum and systematic adjustments of key parameters such as HRT and OLR were investigated. These improvements optimized the contact between microorganisms and substrates, increasing the efficiency of organic contaminant removal with Chemical Oxygen Demand (COD) removals exceeding 67% under all tested conditions, and achieving stable biogas production with high CH₄ content (72-80%) [122].

[121] compared three types of anaerobic reactors for wastewater treatment with high starch content: an Anaerobic Digester (AD), a UASB reactor, and an IC reactor, evaluating them in terms of energy supply and economic factors associated with biogas production. The results showed that both UASB and IC reactors are more energy-efficient than the AD, with the ability to produce excess energy when the Chemical Oxygen Demand (COD) of the wastewater exceeds 15,000 mg/L. The IC reactor achieves an energy and economic balance at COD concentrations of 9,844.67 mg/L and 11,561 mg/L, respectively, which are lower than the concentrations required by the UASB to reach similar balance points. This suggests that the IC reactor is more suitable for efficient and cost-effective treatment of high organic load wastewater.

[123] performed an optimization and model-based analysis on an IC reactor with a total liquid volume of 1963 m³ and gas volume of 213 m³, they analyzed influent sulfur and Influent pH and dissolved CO₂, where sulfur does not have an enough negative impact on energy recovery, handling higher sulfur load only if it is increased the amount of sodium hydroxide keeping the pH above 6.8.

Default conditions produce 135 MWh/d of electricity and heat. In contrast, a lower sulfur influent increased 5.7%, while an unmodified sulfur load reduces only 1.3% CH₄ production and energy recovery.

3.4 Anaerobic Membrane Bioreactors (AnMBR)

The key feature of Membrane Bioreactors (MBRs) is their integrated membrane system, which separates biogas from the liquid phase and solids. This system continuously filters the digested liquid while retaining solids and microorganisms within the digester [124]. This separation occurs in a membrane unit, including microfiltration, ultrafiltration, or nanofiltration [125]. Continuous separation of solids and liquids enables more efficient and prolonged digestion, potentially increasing CH₄ production and process stability [126].

In a study conducted by [97], the operation of an anaerobic membrane bioreactor (AnMBR) treating municipal wastewater in an area with saltwater intrusion was improved by optimizing the OLR and managing high salinity conditions. Increasing the OLR from 1 to 2 kg COD m³ d⁻¹ significantly boosted biogas production, especially with adding methanol or fermented cellulosic sludge. However, adding fermented sludge led to significant membrane fouling, which was mitigated through cleaning with citric acid. Despite adverse saline conditions, membrane operation remained stable, and the treated effluent met EU quality standards for reuse in agriculture.

A study performed on a semi-industrial scale AnMBR (demonstration scale) from a wastewater treatment plant, with a volume of 40 m³, connected to three 0.8 m³ membrane tanks, operated during 610 days, was evaluated during five periods, where the first period achieved the highest CH₄ yield 169.0 ± 95.1 CH₄·kg⁻¹ COD [127].

3.5 Covered Anaerobic Lagoon (CAL)

It is a type of digester that uses a lagoon covered with a membrane to capture biogas produced during the AD of organic matter. The membrane is designed to collect the biogas at the top of the lagoon, creating a gas layer that can be directed to a storage system. They are attractive because of their low construction and operation costs and low capital and operating expenses [128]. However, they are highly dependent on local climatic conditions, and the absence of agitation and sludge accumulation is a primary cause of low energy yield and reduced organic matter removal rates [129, 130].

[131] demonstrates the effectiveness of a polyethylene-CAL reactor for biogas production from poultry rendering wastewater. With a capacity of 15 million gallons, the CAL optimized CH₄ production under natural conditions, achieving up to 80% CH₄ content in the biogas. The lagoon reactor showed a CH₄ production rate of 0.0478 grams per gram of COD for poultry wastewater. The system's capacity to capture about two million kilograms of CH₄ annually offers significant potential for renewable energy generation and reducing greenhouse gas emissions by up to 50,000 tons of CO₂ equivalents per year.

4. Biogas to Biomethane: Purification Technologies and Applications

Raw biogas is composed of two main components, first CH₄, which typically comprises around 50-80% and CO₂, which is present at approximately 20-50% and lower amounts of moisture (H₂O)

and nitrogen (N_2), constituted of about 1-6% and 0-5% respectively. Other compounds present in small amounts are hydrogen sulfide (H_2S), ammonia (NH_3), oxygen (O_2), siloxanes, and volatile organic compounds (VOCs) [132, 133]. Upgrading raw biogas is essential for its effective utilization. This process involves two main steps: the first step is in charge of removing harmful and toxic components that affect CH_4 concentration and biogas quality, whereas the second step improves the calorific value of biogas, which makes its energy accessible to be used as a fuel. It is known that biogas utilization in multiple applications must have a 90% of CH_4 of its total volume, and when it reaches at least a 95% of CH_4 concentration, it is called biomethane [9, 10]. The requirement of purification processes resides on the negative impacts that all pollutant components have, which make it a low-quality biogas to be used as a fuel or energy source, each element present can cause different consequences such as reducing the calorific value, being harmful or toxic, a corrosion or rust source cause, a malfunction cause for engines or clogging the equipment [132].

During the last decade, the issue of purification and upgrading technologies has been extensively explored by several reports and studies, which cover a generous variety of techniques, comparing their positive outcomes and the challenges that come with their application, which are shown in Table 2. These technologies are classified roughly into two main categories: *in situ*, where the upgrading process is done in the same reactor where the biogas is produced, and *ex-situ*, where the upgrading process is done in a different reactor, such as sorption, separation techniques, and some biological methods. There is a third classification for these technologies called hybrid upgrading, which combines *in-situ* and *ex-situ* approaches [134, 135]. In literature, the most addressed and common methods to purify and upgrade biogas include water scrubbing, chemical and physical adsorption, chemical absorption, cryogenic and membrane separation, and biological methods [11, 136]. The following sections will discuss these three categories and their specific techniques in more detail.

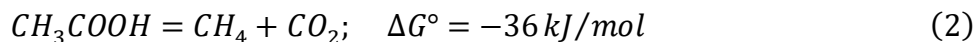
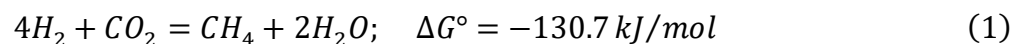
4.1 In-Situ Technologies

In-situ technologies are characterized by their application directly within the anaerobic digester or reactor where the biogas is produced, unlike other technologies that operate in separate locations from the biogas production site [134]. A particular aspect of *in-situ* technologies is that they seem feasible in application for small and medium-scale biogas facilities, in comparison to *ex-situ* technologies, which are more economically and energetically feasible for large-scale biogas producing plants [135]. These *in-situ* technologies can be divided into two main approaches for upgrading biogas: physical/chemical methods and biological methods.

4.1.1 Biological Upgrading

Biological approaches for *in-situ* upgrading are possible due to the presence of organisms that participate in the process of methanogenesis, present inside the biodigester which are capable of converting CO_2 into CH_4 . It relies on methods that make use of hydrogenotrophic methanogenesis (HM), acetoclastic methanogenesis, such as H_2 addition, and electro-methanogenesis [136].

Hydrogen addition is a known method that directly injects H_2 into the biodigester to react with CO_2 , which is the second dominant compound of biogas, this reaction is possible by the hydrogenotrophic (Eq. 1) and acetoclastic (Eq. 2) methanogenesis [137, 138]:



A comparison between both methanogenic pathways showed that hydrogenotrophic methanogenesis is more favorable and stable thermodynamically than acetoclastic methanogenesis [136]. A major advantage of this *in-situ* technique is that it allows the biogas plants to make use of H_2 , which discards the necessity of hydrogen storage; nonetheless counterparts of this technique exist, such as the pH increase by CO_2 consumption, that results in the inhibition of methanogenesis, H_2 can inhibit methanogenesis too, because HM requires a deficient concentration to preserve this CH_4 thermodynamical feasibility [135, 137]. For that reason, to acquire a 99% recovery, completely monitoring and controlling the operational parameters are required [138].

Electro-methanogenesis is a technique that produces CH_4 by reducing CO_2 by direct electron transfer (DET), interspecies electron transfer (IET), and homoacetogenesis paired with acetoclastic methanogenesis. This process requires a bioelectrochemical system, where an electrical current is applied between two electrodes, an anode and a cathode. On IET the H^+ waits to be reduced to H_2 and afterwards a hydrogenotrophic methanogen captures it to produce CH_4 . Alternatively, to DET, where the H^+ produced on the cathode is used directly by methanogens to reduce CO_2 into CH_4 . Lastly, H_2 and CO_2 are converted into acetate by homoacetogenic bacteria. Afterwards, the produced acetate is degraded into CH_4 by acetoclastic methanogenesis [134, 135].

4.1.2 Physical/Chemical Upgrading

The physical/chemical approach for *in-situ* upgrading involves modifications on the pressure parameters, digestate recirculation and the use of additives on the biodigester such as ash, biochar and zero valent iron (ZVI). High pressurization inside the reactor enhances the dissolution of CO_2 in the liquid phase, leading to a gas phase rich in CH_4 when the effluent exits the reactor. This can achieve a CH_4 concentration of 95% or higher. Recirculation of digestate via aerated methanation is also a technique that takes advantage of the different solubility between CH_4 and CO_2 , where changes in pH and temperature can be involved in the solubility of CO_2 . Various aerated reactors resulted in an increase of CH_4 production, with minor washout of anaerobic microorganisms present, and the inhibition of anaerobic populations was observed leaving the tank in the effluent [135, 137]. Lastly, the additives in the case of ash supplementation can act as a booster on AD because of the trace metal elements present in its composition that are essential components of enzymes and cofactors during CH_4 production. Biochar surface area, ash content and oxide metal concentrations, seem attractive to act as cofactors and enhance the CH_4 production consequently. Additionally, the porosity can absorb CO_2 , and microorganisms can attach to the surface area, which prevents them from washing out and helps the direct interspecies electron transfer (DIET). Finally, ZVI provides iron as a cofactor and promotes DIET as well [135].

4.2 Ex-Situ Technologies

Ex-situ technologies are characterized by the application after the biogas extraction from the anaerobic digester; for this reason, they require additional equipment and materials to perform the biogas upgrading, and the consideration of periodic replacement of membranes or solutions for

their correct functionality [137]. *In-situ* technologies focus on reducing the amount of CO₂ to CH₄, to increase its final concentration; on the other hand, *ex-situ* technologies look for the removal not only of CO₂, but also of other impurities such as H₂S, NH₃, and other compounds present in biogas except for CH₄. Additionally, *ex-situ* technologies only become feasible when they are applied to larger-scale biogas plants that exceed a production of 2300 m³ biogas/day [135-137]. These technologies are divided into two categories, sorption technologies and separation technologies. Some examples of sorption technologies are water scrubbing, chemical scrubbing, or pressure swing adsorption, and for separation, examples are membrane and cryogenic separation [9, 139].

4.2.1 Sorption Technologies

Sorption technologies operate on the principles of absorption and adsorption to remove unwanted components from biogas using appropriate materials or solvents. This process helps to ensure that the upgraded biogas exits with minimal traces of pollutants [136]. Absorption technologies are divided into two categories, physical and chemical absorption. For example, water scrubbing is a widely applied physical method for upgrading biogas [140]. On the other hand, adsorption technologies are divided into pressure swing adsorption (PSA), temperature swing adsorption (TSA), and electrical swing adsorption (ESA); PSA, for example, uses pressure to adsorb unwanted components with the support of an absorbent material [9, 141, 142].

Absorption. The absorption relies on the solubilization of gas components on the liquid solvent, thus it takes advantage of CO₂ and H₂S being more soluble than CH₄, in this manner, the gas portion leaving the column is rich in CH₄, reaching concentrations higher than 97% and the liquid leaving the column has high concentrations of CO₂ and H₂S [143, 144]. This technique is divided into physical and chemical absorption. Physical absorption makes use of solvents such as water and organic solvents for scrubbing undesired compounds, high pressure water scrubbing (HPWS) is a widely applied method that utilizes water as solvent, biogas comes from below the column and water goes counter-currently, thus solubilized CO₂ and H₂S leaves from below and upgraded biogas leaves from the top of the column. During this procedure, the biogas gets recirculated to get the majority of upgraded biogas [10, 143]. On the other hand, organic pressure scrubbing (OPS) utilizes different types of solvents such as methanol or polyethylene glycol ethers to solubilize CO₂, H₂S, O₂, N₂, and VOCs because of their higher solubility in comparison to CH₄. Nonetheless, a previous removal of H₂S is adequate [143, 144].

Some disadvantages of absorption technologies applications are that HPWS necessity of high amounts of water, H₂S corrosion if prior removal isn't done, clogging by bacterial growth, high energy consumption for pressure pump and solvent regeneration in OPS, the solvents used in OPS result expensive, solvent regeneration in OPS can be difficult because of H₂S presence, and many others [143].

Adsorption. Adsorption technologies use adsorbent materials with a high surface area, such as activated carbon and alumina; zeolite and silica gel are commonly used to absorb CO₂, N₂, O₂, H₂S, and siloxanes. A PSA process is done on a vertical column and is composed of four steps: adsorption, depressurization, desorption, and pressurization. The adsorption of the compounds is performed when the column is pressurized, after the adsorbent materials get saturated, the upgraded biogas leaves and the biogas is moved to another column, then the saturated column starts to depressurize

to regenerate the adsorbents, till the desorption of the compounds is done and starts a pressurization of the column again to be ready for use. This process depends on multiple columns in continuous operation. TSA and ESA maintain the same steps. Nonetheless, TSA requires thermal energy to regenerate the adsorbents, and ESA requires passing electrical energy to regenerate the adsorbents [9, 144]. Adsorbent materials need to be considered during the selection of the technology, and removal of H₂S is suggested, because its adsorption is normally irreversible and needs to be removed before a new biogas injection is supplied to the PSA column [9, 138].

4.2.2 Separation Technologies

Separation technologies consist of membrane and cryogenic separation; membrane separation, widely used on multiple industrial sectors, functions by permeation which can be performed by gas-gas separation or gas-liquid separation, this technology involves the use of different types of membranes [13, 139, 140]. While cryogenic separation is based on the difference of condensation and boiling points between unwanted biogas compounds and CH₄, this process is performed under low temperatures reaching on the last stage to -85°C and during the cryogenic process pure CO₂ can be sold as a byproduct [13, 136, 144]. Cryogenic separation comes with certain drawbacks as seen on Table 3 which compares advantages and disadvantages of several technologies.

Table 3 Principles advantages and disadvantages for purification and upgrading technologies.

Purification/Upgrading technology	Advantages	Disadvantages	References
<i>In-situ</i>			
H ₂ addition	• Renewable source of H₂ make the entire process renewable • Fully controlled operational parameters could lead to 99% CH₄ recovery	• Limited to lab scale • CO₂ consumption increases pH which inhibits activity of hydrogenotrophic methanogens • Increased H₂ partial pressure inhibits methanogenesis • A high dose of sludge ash could suppress methanogens	[135, 138]
Additive supplementation	• Ashes work as an AD booster • Biochar can adsorb CO₂ • Biochar promotes DIET pathway • Additives could increase CH₄ content	• Biochar could slightly increase pH • The CH₄ final concentration does not meet the biomethane applications' requirements	[135]

Electro-methanogenesis	• Can be performed at low temperatures • Low sludge production • No aeration cost • Renewable electricity sources can be used for H₂O electrolysis • It was observed that 95% CH₄ enrichment	• Limited to lab scale • Still in development	[134, 135, 137]
Ex-situ			
Water scrubbing	• Can reach 99% CH₄ concentration • Less than 2% loss of CH₄ • Feasible at larger scales • Can be useful to remove H₂S • Simple design process • 98-99% CH₄ concentration • Dry process, it does not require water • Can separate N₂, O₂ and CO₂ at the same time	• Requires high amounts of water • Moisture addition to • Low energy efficiency • High investment • High operational costs • Operational problems can be caused by H₂S, corrosion or odor nuisance	[143, 144]
PSA, TSA & ESA	• Equipment compactness • Low capital investment • Easily scalable • 90-98% CH₄ concentration • Low operating and capital costs • Low energy demand • Simple operation • Easily scalable	• NH₃ and H₂S gases block active sites of the absorbent material • 2-4% CH₄ loss • Off-gassing may be prejudicial to the environment because of the CH₄ content • Complexity of its process • H₂S can negatively affect membrane performance • Membranes are expensive • Membrane replacement within 1 to 5 years	[10, 132, 140, 141]
Membrane separation			[13, 136, 141, 143]

Cryogenic separation	• 97% CH₄ concentration • Liquified CO₂ can be commercialized • High removal of impurities	• High energy consumption • Complex process • High operational costs • Requires multiple equipment [10, 13, 136, 140, 143] • Frozen CO₂ can clog the equipment • Requires prior removal H₂O, H₂S and siloxanes • Still in development
<i>Ex-situ</i> H ₂ addition	• Better stability than <i>in situ</i> • Simpler process because of no organic substrate degradation • Non-biomass dependent • 79-98% CH₄ content • Reaches >97% CH₄ concentration	• Technical challenges • Low gas-liquid mass transfer rate [138] • Still in development
Photosynthetic method	• Environment-friendly alternative • Microalgae biomass can later be used as feedstock for biofuel • Can be used for biogas production and wastewater treatment	• Closed systems require high investment • High energy requirements • Possible suppression of microalgae growth by the presence of sulphate [132, 136] • Still in development

Membrane Separation. Membrane technologies work as a permeable barrier that allows desired compounds to pass through; it exploits the differences in properties between the gases in biogas. Simple membrane separation can obtain a 92-94% CH₄ concentration, but the highest achievable requires high-pressure conditions and multi-step gas-gas units, to reach a 96-98% CH₄ concentration [142, 143]. It doesn't need many operational requirements, is low energy consuming and does not represent a difficulty in the installation process; however, it needs a high initial investment which is related to the type of membrane selected, additionally, it requires prior removal of H₂O, H₂S and NH₃ to prevent corrosion [136, 142]. This technique is divided into three categories depending on which membrane is used; inorganic membranes or polymeric membranes, the most commonly used

for raw biogas upgrading, and mixed matrix membranes, polymeric membranes being the preferred because of low cost, simple to produce, and stable during separation processes at high pressure [136].

Cryogenic Separation. Cryogenic separation operates based on differing condensation and boiling points of various biogas constituents. The technique exploits the significant differences in boiling temperatures between CH₄ and other components. For instance, methane boils at -161.5°C, while CO₂ boils at -78.2°C [7, 143]. This technology can achieve CH₄ concentrations up to 97%. However, it is still under development. Cryogenic separation has high energy requirements compared to other technologies, which increases the difficulty of its application. Moreover, it is necessary to do a prior removal of H₂S, siloxanes, water, and halogens to prevent any operational problems in the equipment caused by crystal formation [10, 13, 144].

4.2.3 *Ex-Situ* Biological Upgrading

Such as *in-situ* H₂ addition, the *ex-situ* H₂ addition involves upgrading biogas by using hydrogen produced through the hydrolysis of H₂O, which can be powered by renewable energy sources such as wind or solar. However, in this case, the process occurs in a separate anaerobic reactor containing hydrogenotrophic cultures that convert CO₂ into CH₄. This approach can potentially achieve CH₄ concentrations of up to 95% [9, 141].

Photosynthetic methods are also gaining attention, due to the utilization of microalgae for the design of technologies turning around the sequestration of CO₂ as its biomass. This process can be performed under enclosed or open photobioreactors. Nonetheless, the high investment and energy demand are the most significant drawbacks of this method. The presence of H₂S acts as a suppressor for *Chlorella sp.* performance, a commonly utilized microalgae, which can be handled by the presence of sulphate-oxidizing bacteria and dissolved oxygen, offering a solution for a possible inconvenience for these microalgae [136, 138, 141].

4.3 Biomethane Applications

Biomethane has a remarkable resemblance to natural gas, for that reason, it can be directly injected into gas grids, taking advantage of already existing infrastructure for transporting gas in comparison of H₂, which would require the construction of transport networks, this advantage provides the opportunity to give it a domestic use on stoves and boilers [9, 13, 132]. It is also a promising alternative fuel for vehicles and public transport, it was reported as an effective solution to decrease well-to-wheels (WtW) emissions of compressed natural gas (CNG) buses, by replacing their fuel to compressed biomethane (CBM), and not only that, liquefied biomethane (LBM) was also listed as a preferable fuel for marine and aviation vehicles, due to its high energy density, LBM also has been seen as a possible approach for WtW emissions for heavy duty transport [14, 15, 145]. Biomethane used for electricity generation is applied to combined heat and power (CHP) units and power cells; in this manner, biomethane can be used as a fuel for power plants when the demand is high and renewable energies are not flexible enough. Biomethane, as well, can be efficiently stored, unlike electricity [7, 9, 146]. There are still drawbacks for the application of biomethane to produce energy, because upgrading biogas to biomethane using H₂ only reaches 60% of efficiency, and 36% efficiency when used for electricity production, not only that, several upgrading

technologies result in an economic impact on energy consumption, solvent wastes, operational costs and more aspects that with further investigation will make it more feasible [9].

5. Biomethane Technologies in the World

Biomethane production varies significantly across different world regions, as shown in Figure 2. According to the International Renewable Energy Agency (IRENA) [147] report, the principal countries with better biomethane production are. North America, particularly the United States, has seen substantial growth in biomethane output, being one of the major biomethane producers in the world, driven by favorable policies and abundant feedstock, such as municipal solid waste [148]. Europe leads globally, with countries like Germany, Italy, and Sweden at the forefront, supported by strong renewable energy targets with more than 55,000 GWh annually. Latin America shows promising potential using biomethane, with Brazil making notable strides in biomethane from sugarcane waste [149]. Asia's production is rapidly expanding, led by China, which leads the region's biomethane production, driven by its ambitious goals to reduce air pollution and decrease reliance on imported natural gas [150]. The country has been rapidly developing biogas plants, with a focus on utilizing agricultural waste and municipal solid waste. Chinese policies strongly support biomethane as part of their broader renewable energy strategy. China is one of the countries with a big biomethane production [151]. Africa's biomethane sector is still nascent but growing, with projects emerging in Kenya and South Africa [152]. The Department of Industry, Science, Energy and Resources [153] reports that in Oceania, primarily Australia, is gradually increasing its biomethane production, focusing on landfill gas and agricultural waste as key sources. Biomethane technology as a renewable energy source is increasingly growing worldwide to harness waste and transform it into energy.

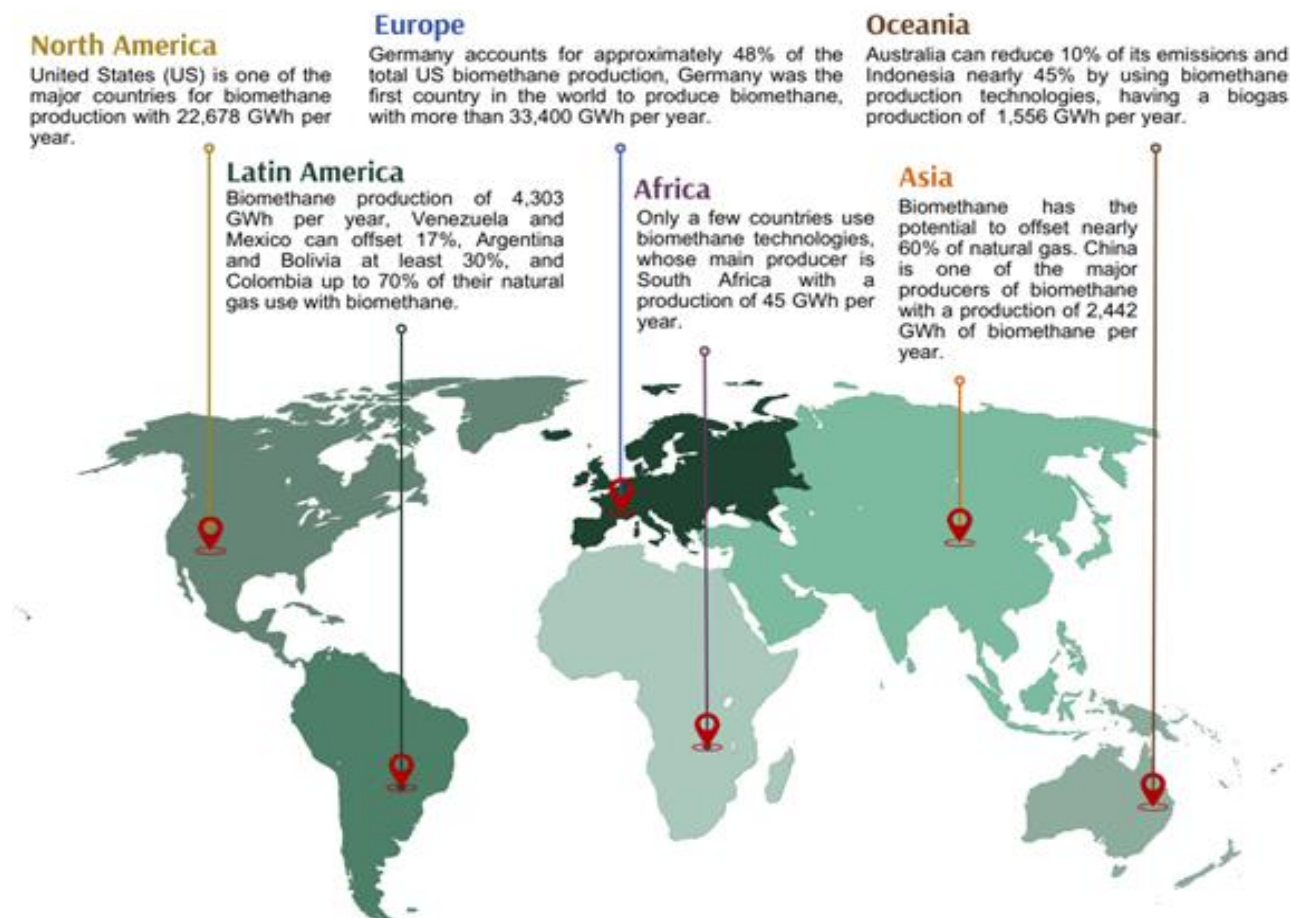


Figure 2 Biomethane Production through AD technologies around the world.

6. Circular Economy

Fossil fuels have facilitated numerous technological developments and have been a crucial resource for everyday life since their discovery. However, due to the significant contribution of greenhouse gas emissions and the anticipated scarcity of fossil fuels, it has prompted the creation of sustainable alternatives for large-scale biofuel production and the interest of governments in many countries to invest in biorefinery infrastructure and research [154].

Biomethane is an essential renewable chemical towards a circular economy. Produced through AD of biomass, it can be used as a compressed local renewable gas to power the public grid or as a vehicle fuel. Full utilization of its potential could cover around 20% of the current global gas demand, leading to a significant reduction of CO₂, CH₄, and other greenhouse gas emissions of up to 80% [155]. Furthermore, the generation of biomethane from effluent treatment or other recycled waste enhances waste management practices, thereby generating a positive impact on social responsibility [156]. Lignocellulosic biorefineries are a foundational element of the circular economy model, as they facilitate the sustainable utilization of natural resources and organic residues, thereby reducing waste generation and transforming these materials into energy sources, biofuels and value-added products such as bioactive compounds (e.g., xylitol, lactic and citric acid, natural red pigment, xylooligosaccharides), biopolymers, microbial pigments, industrial biocatalysts, bioplastics, biosurfactants, biofertilizers and control agents [156-158]. The biogas obtained can be employed directly in heating and fuel for engines. Similarly, CH₄ and CO₂, can serve as effective

precursors for syngas production via several pathways, primarily via reformation, which is used in the production of industrial chemicals (ethanol, dimethyl ether, NH_3 , H_2 , gasoline, diesel, resins, plastics, pharmaceuticals, paints, adhesives, and pressurizing agents, among others) [159]. Moreover, other substances, such as methanol, can be transformed directly without external hydrogen sources through bi-reforming. Conversely, the proportion of CO_2 functions as a precursor to methanol production through [160], and via biological upgrading processes, can be transformed into butanol, ethanol, acetate, and succinate [136].

The transformation of biogas into CH_4 can be converted into electricity and heat, thereby making it a viable option for power generation in cogeneration plants (Figure 3). Furthermore, its use as a fuel for natural gas vehicles contributes to the reduction of emissions in transportation. After purification, biomethane can be injected into the natural gas grid, thus integrating it effectively into the existing energy infrastructure [161, 162]. Additionally, biomethane is a feedstock for manufacturing fertilizer products, primarily NH_3 and urea [163].

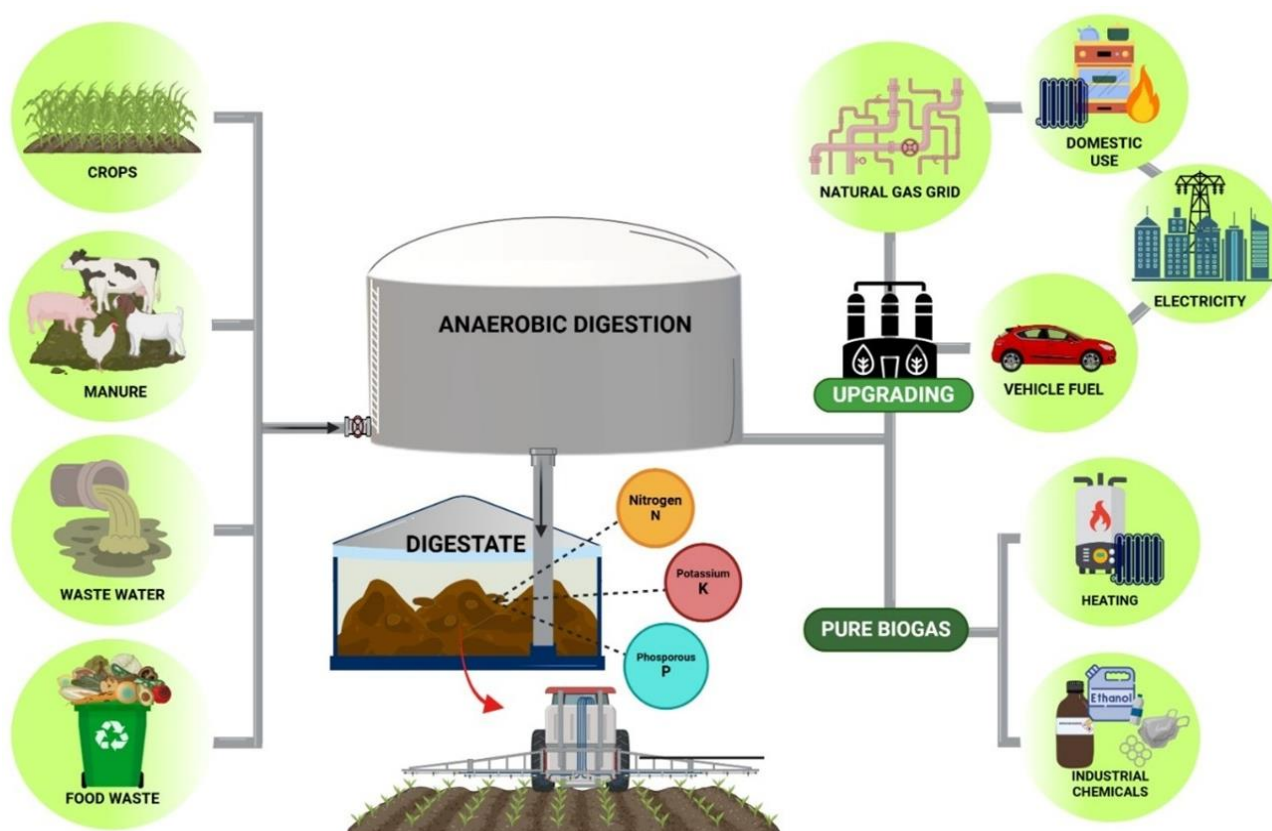


Figure 3 Integration of AD into circular economy, including primary feedstocks and biomethane applications.

Additionally, industrial waste is reintegrated into the value chain, becoming a source of raw materials for other products [164]. The most common is the use of digestate. In AD, about 30% of carbon is transformed into digestate residues, which can be stored and injected into the soil, providing a valuable nitrogen source, such as ammonium nitrogen, which is more readily available for crops than fresh manure or compost [165]. This approach, which utilizes all products and residues throughout the biomass lifecycle, aligns with the principles of the circular economy. Aiming to ensure the efficient and sustainable use of natural resources, replaces the linear industrial model

where materials have a fixed lifespan, and promotes economic growth [166]. It is noteworthy that the global yield of lignocellulosic biomass is estimated to be approximately 181.5 billion tons annually, generating 7 billion tons of residual lignocellulose, the world's most abundant feedstock. However, of the residual biomass produced, only 25% is used intensively [167, 168]. The production of biomethane has the potential to facilitate the advancement of more circular practices within the bioeconomy, since it has been demonstrated that upgrading of biogas is widely profitable and applicable. For effective implementation, it is crucial to collaborate with multiple sectors, including waste management, energy generation, policy making, regulation, infrastructure, and education. Furthermore, it is essential to assess various aspects, such as process dynamics, benefits in energy generation, and environmental benefits [169]. It is also vital to calculate the necessary revenues to ensure the viability of the investment and operation of the treatment and energy recovery systems [142].

The cost of biomethane production oscillates from 0.54 to 0.78 €/m³. Still, it varies according to the plant size, extending from 0.22 to 0.88 US\$ /m³ for plants with a capacity for more than 500 m³/h, and from 1 to 1.55 US\$ /m³ for plants with a capacity of about 100 m³/h. Also, the profitability changes according to the substrate used, and the policy scenarios, such as the type of incentives and the value of the incentive, selling price of biomethane and energy certificates [169, 170]. The initial costs of establishing a biorefinery include investment in infrastructure, such as anaerobic digesters, and biogas treatment equipment. However, as the plant reaches operating capacity, costs per unit of CH₄ tend to decrease due to economies of scale [171]. Some studies estimate that over the next 30 years the cost of production is expected to decline by around €57-66/MWh, assuming a similar increase in the scale of production [142]. Bose and collaborators assessed the co-generation of biomethane, food (protein supplement), and biofertilizer through photosynthetic upgrading of biogas on different scales. They determined that at an industrial scale, a polygeneration plant could be profitable without incentives, and the cost of final value-added products was competitive with market alternatives across all scales. However, in medium and small-scale systems, higher biomethane prices are required to be viable [172].

Since the production of biomethane has been demonstrated as a promising alternative to replace fossil fuels, the interest in many countries has been directed to implementing sustainable technologies to decrease environmental impact and simultaneously achieve economic profit in a circular economy model. European countries, China, and the United States are responsible for over 90% of worldwide production [17]. In Europe alone, from 2018 to 2020, the number of biogas plants increased by 50% in 18 countries compared to the last decade, with Germany being the country with the highest number of plants and the largest producer, followed by the United Kingdom, Italy, and France [7].

Through a techno-economic analysis conducted by [129], focused on biomethane-urea co-production from biogas in different plant sizes in four European countries (Spain, Italy, Germany, and the United Kingdom), it was demonstrated that a robust subsidy policy is a crucial element in attaining profitability at the small-scale level. Nevertheless, large-scale plants with a capacity of 500-1000 m³/h, operating synergistically to produce biomethane and urea in Italy and Spain, are regarded as more profitable even with relatively minimal subsidies.

In a study by [140], the potential of biomethane as an alternative fuel source for heavy and light vehicles, as well as for domestic applications, was evaluated in Brazil. The researchers reported that diesel substitution was most beneficial for heavy-duty vehicles, where it demonstrated the highest

superior environmental performance in the other categories evaluated. Therefore, biomethane technologies for automobiles present an opportunity to establish a cleaner energy matrix in Brazil. However, it is still necessary to improve technologies and optimize processes to significantly reduce emissions of other pollutants by substituting liquefied petroleum gas (LPG) in domestic use and gasoline in light vehicles with biomethane [173]. It is worth noting that Brazil has the highest number of patents for technological advances in using vinasse for biomethane production [149].

Nonetheless, it is essential to consider that AD can release air pollutants during the processes before and after AD, among them are nitrogen oxides, sulfur dioxide, carbon monoxide, CO₂, CH₄, and black carbon, raising concerns about the environmental risks associated with the process [174]. For this reason, some measures to prevent these gases from leaking into the atmosphere are to cover tanks, using a flare to avoid CH₄ discharge, and improve the efficiency of CHP units and electricity power utilization strategies [175]. Other aspects to consider in GHG emissions related to AD are the acquisition of substrates, the pretreatment required, their storage and transport, and the utilization of all products that can be obtained through AD. A study done by [176], analyzed 30 AD plants in France and the evaluation of their different scenarios stated that AD plants do not significantly benefit to GHG emissions mitigation, considering substrate production, biomass storage and transport, AD process, fossil fuel substitution, composting process, biomass application to soils among other details to offer a reproducible methodology to evaluate AD plants. Because of this, it is necessary to develop further research and perform rigorous evaluations related to biogas production. For example, [177] used *Sargassum* for biogas generation, initially *Sargassum* disposal has the potential to generate 0.33 CO₂ eq./kg *Sargassum*, however, through a techno-economic and environmental impact assessment, where four different scenarios were evaluated, considering harvesting till the end-use, two of them added food waste and achieved a lower the generation of CO₂, obtaining results between 0.005 to 0.042 CO₂ eq./kg *Sargassum*/food waste. In another research, the environmental impact of compressed biogas production using manure and municipal organic waste separately was evaluated, showing that compressed biogas production can potentially lower the effects of climate change; however, acidification and eutrophication were adverse outcomes [178].

Additionally, pretreatment steps also hinder the environmental impact. For example, chemical pre-treatment GHG emissions are generally low. However, improper management could lead to acidification or eutrophication. Mechanical methods are high-energy-consuming and could release SO₂ and NO, causing acidification, while thermal methods are also energy-demanding. On the other hand, biological methods can be an environmentally friendly pretreatment, due to lower energy requirement, lower GHG emissions rate, and lesser generation of hazardous residues [179].

Many of the issues around the environmental impact of AD are related to harmful practices throughout the production process. Thus, it is required to establish regulations and proper management of the substrate, products, and byproducts. Biomethane can be a promising renewable energy source; however, it is still required to address specific topics around the process to contribute to lowering climate change.

Moreover, implementing a circular economy model not only aims to reduce waste generation but also to add value to waste for as long as possible. It also aims to achieve economic, environmental, and social benefits. This model must reduce pressure on the environment by promoting the reuse of resources, which would contribute to a more sustainable management of natural resources [180]. Furthermore, the diversification and utilization of all organic waste should

enhance the security of supply of raw materials. From an economic standpoint, the transition to a circular model should stimulate economic growth by creating new markets and business opportunities, as well as generate employment by opening new biogas and biomethane production facilities and jobs related to feedstock processing and the production process [181, 182]. Finally, for companies and governments to adopt this model on a large scale, there must be an economic return on investment that offsets the initial costs and investments required in research and infrastructure. This financial return would not only provide sufficient motivation to adopt the circular model. However, it would also ensure that the transition to a more sustainable economy is attractive and financially viable for businesses and governments [183].

7. Conclusion

Biomethane production holds significant potential due to its cost-effectiveness and environmental benefits. By converting organic waste, such as crop residues, manure, and wastewater, into biogas, the process not only reduces greenhouse gas emissions but also improves waste management and offers economic advantages. To maximize CH₄ yields, innovations in AD design, feedstock pretreatments, co-digestion, DIET-based strategies, and biogas upgrading systems are essential. These advancements will facilitate integrating biomethane production into urban and industrial settings, decreasing reliance on non-renewable energy sources. Currently, emerging strategies are mainly focused on DIET enhancement, which is revolutionizing AD by making it faster, more efficient, and more resilient. Conductive materials, targeted enrichment of DIET microbes, optimized reactor designs, and advanced monitoring technologies form a comprehensive approach to harness DIET's full potential. However, transitioning from empirical observations to a mechanistic understanding is essential for advancing the field. Future research should prioritize quantifying DIET's specific contribution to CH₄ production, developing reliable biomarkers to monitor its activity, and tailoring conductive materials to match the needs of distinct microbial communities and feedstocks. Focusing on these challenges will be key to unlocking the scalability and reliability of DIET-enhanced AD systems for sustainable waste treatment and renewable energy generation.

As the core technology in the biomethane industry, AD leverages the potential of organic waste as feedstock, positioning it as a tool for reducing environmental impact while promoting sustainability. Biomethane production not only contributes to reducing emissions but also offers substantial economic benefits by converting waste into valuable products. The efficiency of this process, however, heavily depends on optimizing operational parameters such as reactor design, OLR, and HRT.

Future advancements in reactor technology should focus on improving process stability and efficiency. This can be achieved by refining reactor designs, incorporating real-time monitoring systems, and optimizing operational strategies. At the same time, improvements in feedstock pretreatment and biogas upgrading plants will further enhance CH₄ yields, ensuring that AD systems can be seamlessly integrated into urban and industrial infrastructures to replace traditional fossil fuels.

Purification and upgrading technologies remain crucial for improving biogas quality, making biomethane a viable fuel for various applications such as injection into gas grids, electricity generation, and transportation fuel. However, challenges remain, including CH₄ loss during upgrading processes and the energy consumption of some purification methods. Innovations in

adsorbent materials and energy-efficient technologies will be vital for minimizing these losses and lowering operational costs. While some upgrading methods show promise due to their low environmental impact and high CH₄ purity, further development and scaling are needed before they can be widely adopted.

The similarity of biomethane to natural gas offers significant advantages, positioning it as a potential alternative to conventional fuels. However, to fully harness biomethane potential, challenges in purification and upgrading processes must be addressed. Although still in its early stages, utilizing biomethane as a clean energy source will require further research and technological advances to establish it as a competitive option in energy markets.

Given the current global climate challenges, the need for refineries and industries to adopt the circular economy model has become imperative. This approach extends the life cycle of raw materials, enhances waste utilization, and supports the production of renewable energies. Biomethane production, as part of this model, aligns with the broader goals of sustainability and reducing environmental impact, making it a crucial component in the transition to a cleaner, more sustainable energy future.

Author Contributions

N.B.: idea conceptualization, supervision, edition and revision of the manuscript. G, M-M., L.A.L-G., L.Y.A-M., R.M-G., J.P.S-M. and I.A.C-G.: literature review and draft writing. M.J.C-A., J.D.S-M., M.P.L-E. and AYH-A.: revision of the manuscript.

Competing Interests

The authors have declared that no competing interests exist.

References

1. Blondeel M, Bradshaw MJ, Bridge G, Kuzemko C. The geopolitics of energy system transformation: A review. *Geogr Compass*. 2021; 15: e12580.
2. Lovering J, Swain M, Blomqvist L, Hernandez RR. Land-use intensity of electricity production and tomorrow's energy landscape. *PLoS One*. 2022; 17: e0270155.
3. Ravindran R, Donkor K, Gottumukkala L, Menon A, Guneratnam AJ, McMahon H, et al. Biogas, biomethane and digestate potential of by-products from green biorefinery systems. *Clean Technol*. 2022; 4: 35-50.
4. Dhull P, Lohchab RK, Kumar S, Kumari M, Shaloo, Bhankhar AK. Anaerobic digestion: Advance techniques for enhanced biomethane/biogas production as a source of renewable energy. *Bioenergy Res*. 2024; 17: 1228-1249.
5. Song J, Wang Y, Zhang S, Song Y, Xue S, Liu L, et al. Coupling biochar with anaerobic digestion in a circular economy perspective: A promising way to promote sustainable energy, environment and agriculture development in China. *Renew Sustain Energy Rev*. 2021; 144: 110973.
6. Tyagi VK, Bhatia A, Kubota K, Rajpal A, Ahmed B, Khan AA, et al. Microbial community dynamics in anaerobic digesters treating organic fraction of municipal solid waste. *Environ Technol Innov*. 2021; 21: 101303.

7. Pavičić J, Novak Mavar K, Brkić V, Simon K. Biogas and biomethane production and usage: Technology development, advantages and challenges in Europe. *Energies*. 2022; 15: 2940.
8. Roopnarain A, Rama H, Ndaba B, Bello-Akinosho M, Bamuya-Pemu E, Adeleke R. Unravelling the anaerobic digestion 'black box': Biotechnological approaches for process optimization. *Renew Sustain Energy Rev*. 2021; 152: 111717.
9. Adnan AI, Ong MY, Nomanbhay S, Chew KW, Show PL. Technologies for biogas upgrading to biomethane: A review. *Bioengineering*. 2019; 6: 92.
10. Kapoor R, Ghosh P, Kumar M, Vijay VK. Evaluation of biogas upgrading technologies and future perspectives: A review. *Environ Sci Pollut Res*. 2019; 26: 11631-11661.
11. Swinbourn R, Li C, Wang F. A comprehensive review on biomethane production from biogas separation and its techno-economic assessments. *ChemSusChem*. 2024; 17: e202400779.
12. Jasińska A, Grosser A, Meers E. Possibilities and limitations of anaerobic co-digestion of animal manure-A critical review. *Energies*. 2023; 16: 3885.
13. Franco AC, Franco LS, Tesser DP, Salvador R, de Francisco AC, Picinin CT, et al. Advances and challenges on the technologies and applications of biomethane. *Energy Sources Part A*. 2025; 47: 5786-5796.
14. Noussan M. The use of biomethane in internal combustion engines for public transport decarbonization: A case study. *Energies*. 2023; 16: 7995.
15. Rogala Z, Stanclik M, Łuszkiewicz D, Malecha Z. Perspectives for the use of biogas and biomethane in the context of the green energy transformation on the example of an EU country. *Energies*. 2023; 16: 1911.
16. Ah-Voun D, Chyong CK, Li C. Europe's energy security: From Russian dependence to renewable reliance. *Energy Policy*. 2024; 184: 113856.
17. Marconi P, Rosa L. Role of biomethane to offset natural gas. *Renew Sustain Energy Rev*. 2023; 187: 113697.
18. Rodrigues BC, de Mello BS, Grangeiro LC, Dussan KJ, Sarti A. The most important technologies and highlights for biogas production worldwide. *J Air Waste Manag Assoc*. 2025; 75: 87-108.
19. Srivastava SK. Advancement in biogas production from the solid waste by optimizing the anaerobic digestion. *Waste Dispos Sustain Energy*. 2020; 2: 85-103.
20. Wang Z, Hu Y, Wang S, Wu G, Zhan X. A critical review on dry anaerobic digestion of organic waste: Characteristics, operational conditions, and improvement strategies. *Renew Sustain Energy Rev*. 2023; 176: 113208.
21. Lavergne C, Bovio-Winkler P, Etchebehere C, García-Gen S. Towards centralized biogas plants: Co-digestion of sewage sludge and pig manure maintains process performance and active microbiome diversity. *Bioresour Technol*. 2020; 297: 122442.
22. Krohn C, Khudur L, Dias DA, Van den Akker B, Rees CA, Crosbie ND, et al. The role of microbial ecology in improving the performance of anaerobic digestion of sewage sludge. *Front Microbiol*. 2022; 13: 1079136.
23. Blair EM, Dickson KL, O'Malley MA. Microbial communities and their enzymes facilitate degradation of recalcitrant polymers in anaerobic digestion. *Curr Opin Microbiol*. 2021; 64: 100-108.
24. Mutungwazi A, Ijoma GN, Matambo TS. The significance of microbial community functions and symbiosis in enhancing methane production during anaerobic digestion: A review. *Symbiosis*. 2021; 83: 1-24.

25. Castellano-Hinojosa A, Armato C, Pozo C, González-Martínez A, González-López J. New concepts in anaerobic digestion processes: Recent advances and biological aspects. *Appl Microbiol Biotechnol.* 2018; 102: 5065-5076.
26. Akindolire MA, Rama H, Roopnarain A. Psychrophilic anaerobic digestion: A critical evaluation of microorganisms and enzymes to drive the process. *Renew Sustain Energy Rev.* 2022; 161: 112394.
27. Mu L, Wang Y, Xu F, Li J, Tao J, Sun Y, et al. Emerging strategies for enhancing propionate conversion in anaerobic digestion: A review. *Molecules.* 2023; 28: 3883.
28. Steiniger B, Hupfau S, Insam H, Schaum C. Exploring anaerobic digestion from mesophilic to thermophilic temperatures-operational and microbial aspects. *Fermentation.* 2023; 9: 798.
29. Sillero L, Perez M, Solera R. Temperature-phased enhanced the single-stage anaerobic co-digestion of sewage sludge, wine vinasse and poultry manure: Perspectives for the circular economy. *Fuel.* 2023; 331: 125761.
30. Hu Y, Shen C. Thermophilic-mesophilic temperature phase anaerobic co-digestion compared with single phase co-digestion of sewage sludge and food waste. *Sci Rep.* 2024; 14: 11967.
31. Rocamora I, Wagland ST, Villa R, Simpson EW, Fernández O, Bajón-Fernández Y. Dry anaerobic digestion of organic waste: A review of operational parameters and their impact on process performance. *Bioresour Technol.* 2020; 299: 122681.
32. Rocha-Meneses L, Zannerni R, Inayat A, Abdallah M, Shanableh A, Ghenai C, et al. Current progress in anaerobic digestion reactors and parameters optimization. *Biomass Conv Bioref.* 2022. doi: 10.1007/s13399-021-02224-z.
33. Hamzah AF, Hamzah MH, Man HC, Jamali NS, Siajam SI, Ismail MH. Effect of organic loading on anaerobic digestion of cow dung: Methane production and kinetic study. *Heliyon.* 2023; 9: e16791.
34. Amin FR, Khalid H, El-Mashad HM, Chen C, Liu G, Zhang R. Functions of bacteria and archaea participating in the bioconversion of organic waste for methane production. *Sci Total Environ.* 2021; 763: 143007.
35. Nielsen BV, Maneein S, Al Farid MM, Milledge JJ. The effects of halogenated compounds on the anaerobic digestion of macroalgae. *Fermentation.* 2020; 6: 85.
36. Alrawashdeh KA, Gul E, Yang Q, Yang H, Bartocci P, Fantozzi F. Effect of heavy metals in the performance of anaerobic digestion of olive mill waste. *Processes.* 2020; 8: 1146.
37. Kadam R, Khanthong K, Jang H, Lee J, Park J. Occurrence, fate, and implications of heavy metals during anaerobic digestion: A review. *Energies.* 2022; 15: 8618.
38. Guo Q, Majeed S, Xu R, Zhang K, Kakade A, Khan A, et al. Heavy metals interact with the microbial community and affect biogas production in anaerobic digestion: A review. *J Environ Manage.* 2019; 240: 266-272.
39. Zhang H, Xu Y, Tian Y, Zheng L, Hao H, Huang H. Impact of Fe and Ni addition on the VFAs' generation and process stability of anaerobic fermentation containing Cd. *Int J Environ Res Public Health.* 2019; 16: 4066.
40. Li X, Huang J, Liu Y, Huang T, Maurer C, Kranert M. Effects of salt on anaerobic digestion of food waste with different component characteristics and fermentation concentrations. *Energies.* 2019; 12: 3571.
41. Zupančič GD, Panjičko M, Marinšek Logar R, Lavrič L, Zorec M, Fanedl L. Salinity inhibition in thermophilic anaerobic digestion of organic waste. *Appl Sci.* 2023; 13: 6590.

42. Wang S, Hou X, Su H. Exploration of the relationship between biogas production and microbial community under high salinity conditions. *Sci Rep.* 2017; 7: 1149.
43. Cai Y, Zhu M, Meng X, Zhou JL, Zhang H, Shen X. The role of biochar on alleviating ammonia toxicity in anaerobic digestion of nitrogen-rich wastes: A review. *Bioresour Technol.* 2022; 351: 126924.
44. Capson-Tojo G, Moscoviz R, Astals S, Robles Á, Steyer JP. Unraveling the literature chaos around free ammonia inhibition in anaerobic digestion. *Renew Sustain Energy Rev.* 2020; 117: 109487.
45. Jiang Y, McAdam E, Zhang Y, Heaven S, Banks C, Longhurst P. Ammonia inhibition and toxicity in anaerobic digestion: A critical review. *J Water Process Eng.* 2019; 32: 100899.
46. Shi X, Lin J, Zuo J, Li P, Li X, Guo X. Effects of free ammonia on volatile fatty acid accumulation and process performance in the anaerobic digestion of two typical bio-wastes. *J Environ Sci.* 2017; 55: 49-57.
47. Prem EM, Mutschlechner M, Stres B, Illmer P, Wagner AO. Lignin intermediates lead to phenyl acid formation and microbial community shifts in meso-and thermophilic batch reactors. *Biotechnol Biofuels.* 2021; 14: 27.
48. Prem EM, Schwarzenberger A, Markt R, Wagner AO. Effects of phenyl acids on different degradation phases during thermophilic anaerobic digestion. *Front Microbiol.* 2023; 14: 1087043.
49. He J, Luo T, Shi Z, Angelidaki I, Zhang S, Luo G. Microbial shifts in anaerobic digestion towards phenol inhibition with and without hydrochar as revealed by metagenomic binning. *J Hazard Mater.* 2022; 440: 129718.
50. Braun R. Anaerobic digestion: A multi-faceted process for energy, environmental management and rural development. In: *Improvement of crop plants for industrial end uses.* Dordrecht, The Netherlands: Springer; 2007. pp. 335-416.
51. Weiland P. Biogas production: Current state and perspectives. *Appl Microbiol Biotechnol.* 2010; 85: 849-860.
52. Dar RA, Parmar M, Dar EA, Sani RK, Phutela UG. Biomethanation of agricultural residues: Potential, limitations and possible solutions. *Renew Sustain Energy Rev.* 2021; 135: 110217.
53. Mosier N, Hendrickson R, Ho N, Sedlak M, Ladisch MR. Optimization of pH controlled liquid hot water pretreatment of corn stover. *Bioresour Technol.* 2005; 96: 1986-1993.
54. Poszytek K, Ciezkowska M, Sklodowska A, Drewniak L. Microbial consortium with high cellulolytic activity (MCHCA) for enhanced biogas production. *Front Microbiol.* 2016; 7: 324.
55. Li X, Dang F, Zhang YT, Zou D, Yuan H. Anaerobic digestion performance and mechanism of ammoniation pretreatment of corn stover. *Bioresources.* 2015; 10: 5777-5790.
56. Lehtomäki A, Viinikainen TA, Rintala JA. Screening boreal energy crops and crop residues for methane biofuel production. *Biomass Bioenergy.* 2008; 32: 541-550.
57. Kaur K, Phutela UG. Enhancement of paddy straw digestibility and biogas production by sodium hydroxide-microwave pretreatment. *Renew Energy.* 2016; 92: 178-184.
58. Herrmann C, Idler C, Heiermann M. Biogas crops grown in energy crop rotations: Linking chemical composition and methane production characteristics. *Bioresour Technol.* 2016; 206: 23-35.
59. Himanshu H, Voelklein MA, Murphy JD, Grant J, O'Kiely P. Factors controlling headspace pressure in a manual manometric BMP method can be used to produce a methane output comparable to AMPTS. *Bioresour Technol.* 2017; 238: 633-642.

60. Yong Z, Dong Y, Zhang X, Tan T. Anaerobic co-digestion of food waste and straw for biogas production. *Renew Energy*. 2015; 78: 527-530.
61. Janke L, Leite A, Nikolausz M, Schmidt T, Liebetrau J, Nelles M, et al. Biogas production from sugarcane waste: Assessment on kinetic challenges for process designing. *Int J Mol Sci*. 2015; 16: 20685-20703.
62. Kabir MM, Rajendran K, Taherzadeh MJ, Horváth IS. Experimental and economical evaluation of bioconversion of forest residues to biogas using organosolv pretreatment. *Bioresour Technol*. 2015; 178: 201-208.
63. Isci AS, Demirer GN. Biogas production potential from cotton wastes. *Renew Energy*. 2007; 32: 750-757.
64. Patinvoh RJ, Lundin M, Taherzadeh MJ, Sárvári Horváth I. Dry anaerobic co-digestion of citrus wastes with keratin and lignocellulosic wastes: Batch and continuous processes. *Waste Biomass Valoriz*. 2020; 11: 423-434.
65. Mankar AR, Pandey A, Modak A, Pant KK. Pretreatment of lignocellulosic biomass: A review on recent advances. *Bioresour Technol*. 2021; 334: 125235.
66. Liu M, Wei Y, Leng X. Improving biogas production using additives in anaerobic digestion: A review. *J Clean Prod*. 2021; 297: 126666.
67. Kainthola J, Kalamdhad AS, Goud VV. A review on enhanced biogas production from anaerobic digestion of lignocellulosic biomass by different enhancement techniques. *Process Biochem*. 2019; 84: 81-90.
68. Rasapoor M, Young B, Brar R, Sarmah A, Zhuang WQ, Baroutian S. Recognizing the challenges of anaerobic digestion: Critical steps toward improving biogas generation. *Fuel*. 2020; 261: 116497.
69. Barakat A, de Vries H, Rouau X. Dry fractionation process as an important step in current and future lignocellulose biorefineries: A review. *Bioresour Technol*. 2013; 134: 362-373.
70. Taherzadeh MJ, Karimi K. Pretreatment of lignocellulosic wastes to improve ethanol and biogas production: A review. *Int J Mol Sci*. 2008; 9: 1621-1651.
71. Liu C, Wyman CE. The effect of flow rate of compressed hot water on xylan, lignin, and total mass removal from corn stover. *Ind Eng Chem Res*. 2003; 42: 5409-5416.
72. Kumakura M, Kaetsu I. Effect of radiation pretreatment of bagasse on enzymatic and acid hydrolysis. *Biomass*. 1983; 3: 199-208.
73. Marin J, Kennedy KJ, Eskicioglu C. Enhanced solubilization and anaerobic biodegradability of source-separated kitchen waste by microwave pre-treatment. *Waste Manage Res*. 2011; 29: 208-218.
74. Zeng J, Singh D, Gao D, Chen S. Effects of lignin modification on wheat straw cell wall deconstruction by *Phanerochaete chrysosporium*. *Biotechnol Biofuels*. 2014; 7: 161.
75. Shevchenko SM, Beatson RP, Saddler JN. The nature of lignin from steam explosion/enzymatic hydrolysis of softwood: structural features and possible uses: Scientific note. *Appl Biochem Biotechnol*. 1999; 77-79: 867-876.
76. Mood SH, Golfeshan AH, Tabatabaei M, Jouzani GS, Najafi GH, Gholami M, et al. Lignocellulosic biomass to bioethanol, a comprehensive review with a focus on pretreatment. *Renew Sustain Energy Rev*. 2013; 27: 77-93.
77. Bhatia SK, Kim SH, Yoon JJ, Yang YH. Current status and strategies for second generation biofuel production using microbial systems. *Energy Convers Manage*. 2017; 148: 1142-1156.

78. Okano K, Kitagawa M, Sasaki Y, Watanabe T. Conversion of Japanese red cedar (*Cryptomeria japonica*) into a feed for ruminants by white-rot basidiomycetes. *Anim Feed Sci Technol.* 2005; 120: 235-243.
79. Bumpus JA, Tien M, Wright D, Aust SD. Oxidation of persistent environmental pollutants by a white rot fungus. *Science.* 1985; 228: 1434-1436.
80. Zhong W, Zhang Z, Luo Y, Sun S, Qiao W, Xiao M. Effect of biological pretreatments in enhancing corn straw biogas production. *Bioresour Technol.* 2011; 102: 11177-11182.
81. Mussoline W, Esposito G, Lens P, Garuti G, Giordano A. Design considerations for a farm-scale biogas plant based on pilot-scale anaerobic digesters loaded with rice straw and piggery wastewater. *Biomass Bioenergy.* 2012; 46: 469-478.
82. Shrestha B, Hernandez R, Fortela DL, Sharp W, Chistoserdov A, Gang D, et al. A review of pretreatment methods to enhance solids reduction during anaerobic digestion of municipal wastewater sludges and the resulting digester performance: Implications to future urban biorefineries. *Appl Sci.* 2020; 10: 9141.
83. Karki R, Chuenchart W, Surendra KC, Shrestha S, Raskin L, Sung S, et al. Anaerobic co-digestion: Current status and perspectives. *Bioresour Technol.* 2021; 330: 125001.
84. Yangin-Gomec C, Ozturk I. Effect of maize silage addition on biomethane recovery from mesophilic co-digestion of chicken and cattle manure to suppress ammonia inhibition. *Energy Convers Manage.* 2013; 71: 92-100.
85. Girault R, Bridoux G, Nauleau F, Poullain C, Buffet J, Peu P, et al. Anaerobic co-digestion of waste activated sludge and greasy sludge from flotation process: Batch versus CSTR experiments to investigate optimal design. *Bioresour Technol.* 2012; 105: 1-8.
86. Zhang Y, Banks CJ, Heaven S. Co-digestion of source segregated domestic food waste to improve process stability. *Bioresour Technol.* 2012; 114: 168-178.
87. Karthikeyan OP, Visvanathan C. Effect of C/N ratio and ammonia-N accumulation in a pilot-scale thermophilic dry anaerobic digester. *Bioresour Technol.* 2012; 113: 294-302.
88. Sawatdeenarunat C, Surendra KC, Takara D, Oechsner H, Khanal SK. Anaerobic digestion of lignocellulosic biomass: Challenges and opportunities. *Bioresour Technol.* 2015; 178: 178-186.
89. Ziels RM, Nobu MK, Sousa DZ. Elucidating syntrophic butyrate-degrading populations in anaerobic digesters using stable-isotope-informed genome-resolved metagenomics. *mSystems.* 2019; 4. doi: 10.1128/msystems.00159-19.
90. Baek G, Kim J, Kim J, Lee C. Role and potential of direct interspecies electron transfer in anaerobic digestion. *Energies.* 2018; 11: 107.
91. Thauer RK, Jungermann K, Decker K. Energy conservation in chemotrophic anaerobic bacteria. *Bacteriol Rev.* 1977; 41: 100-180.
92. Li H, Han K, Li Z, Zhang J, Li H, Huang Y, et al. Performance, granule conductivity and microbial community analysis of upflow anaerobic sludge blanket (UASB) reactors from mesophilic to thermophilic operation. *Biochem Eng J.* 2018; 133: 59-65.
93. Portela PC, Shipps CC, Shen C, Srikanth V, Salgueiro CA, Malvankar NS. Widespread extracellular electron transfer pathways for charging microbial cytochrome OmcS nanowires via periplasmic cytochromes PpcABCDE. *Nat Commun.* 2024; 15: 2434.
94. Liu X, Huang L, Rensing C, Ye J, Nealson KH, Zhou S. Syntrophic interspecies electron transfer drives carbon fixation and growth by *Rhodospseudomonas palustris* under dark, anoxic conditions. *Sci Adv.* 2021; 7: eabh1852.

95. Zhang C, Yang R, Sun M, Zhang S, He M, Tsang DC, et al. Wood waste biochar promoted anaerobic digestion of food waste: Focusing on the characteristics of biochar and microbial community analysis. *Biochar*. 2022; 4: 62.
96. Chen Q, Liu C, Liu X, Sun D, Li P, Qiu B, et al. Magnetite enhances anaerobic digestion of high salinity organic wastewater. *Environ Res*. 2020; 189: 109884.
97. Chowdhury B, Lin L, Dhar BR, Islam MN, McCartney D, Kumar A. Enhanced biomethane recovery from fat, oil, and grease through co-digestion with food waste and addition of conductive materials. *Chemosphere*. 2019; 236: 124362.
98. Van Steendam C, Smets I, Skerlos S, Raskin L. Improving anaerobic digestion via direct interspecies electron transfer requires development of suitable characterization methods. *Curr Opin Biotechnol*. 2019; 57: 183-190.
99. Lovley DR. Syntrophy goes electric: Direct interspecies electron transfer. *Annu Rev Microbiol*. 2017; 71: 643-664.
100. Zhao Z, Zhang Y, Yu Q, Dang Y, Li Y, Quan X. Communities stimulated with ethanol to perform direct interspecies electron transfer for syntrophic metabolism of propionate and butyrate. *Water Res*. 2016; 102: 475-484.
101. Hagos K, Zong J, Li D, Liu C, Lu X. Anaerobic co-digestion process for biogas production: Progress, challenges and perspectives. *Renew Sustain Energy Rev*. 2017; 76: 1485-1496.
102. Bhatt AH, Tao L. Economic perspectives of biogas production via anaerobic digestion. *Bioengineering*. 2020; 7: 74.
103. Chen H, Wei Y, Xie C, Wang H, Chang S, Xiong Y, et al. Anaerobic treatment of glutamate-rich wastewater in a continuous UASB reactor: Effect of hydraulic retention time and methanogenic degradation pathway. *Chemosphere*. 2020; 245: 125672.
104. Show KY, Yan Y, Yao H, Guo H, Li T, Show DY, et al. Anaerobic granulation: A review of granulation hypotheses, bioreactor designs and emerging green applications. *Bioresour Technol*. 2020; 300: 122751.
105. Cecconet D, Callegari A, Capodaglio AG. UASB performance and perspectives in urban wastewater treatment at sub-mesophilic operating temperature. *Water*. 2022; 14: 115.
106. Passos F, Bressani-Ribeiro T, Rezende S, Chernicharo CA. Potential applications of biogas produced in small-scale UASB-based sewage treatment plants in Brazil. *Energies*. 2020; 13: 3356.
107. Mainardis M, Buttazzoni M, Goi D. Up-flow anaerobic sludge blanket (UASB) technology for energy recovery: A review on state-of-the-art and recent technological advances. *Bioengineering*. 2020; 7: 43.
108. Mariraj Mohan S, Swathi T. A review on upflow anaerobic sludge blanket reactor: Factors affecting performance, modification of configuration and its derivatives. *Water Environ Res*. 2022; 94: e1665.
109. Wu J, Jiang B, Kong Z, Yang C, Li L, Feng B, et al. Improved stability of up-flow anaerobic sludge blanket reactor treating starch wastewater by pre-acidification: Impact on microbial community and metabolic dynamics. *Bioresour Technol*. 2021; 326: 124781.
110. Guo Q, Yang Z, Zhang B, Hua M, Liu C, Pan B. Enhanced methane production during long-term UASB operation at high organic loads as enabled by the immobilized Fungi. *Front Environ Sci Eng*. 2022; 16: 71.

111. Heydari B, Sharghi EA, Rafiee S, Mohtasebi SS. Use of artificial neural network and adaptive neuro-fuzzy inference system for prediction of biogas production from spearmint essential oil wastewater treatment in up-flow anaerobic sludge blanket reactor. *Fuel*. 2021; 306: 121734.
112. Arthur PM, Konaté Y, Sawadogo B, Sagoe G, Dwumfour-Asare B, Ahmed I, et al. Performance evaluation of a full-scale upflow anaerobic sludge blanket reactor coupled with trickling filters for municipal wastewater treatment in a developing country. *Heliyon*. 2022; 8: e10129.
113. Kariyama ID, Zhai X, Wu B. Influence of mixing on anaerobic digestion efficiency in stirred tank digesters: A review. *Water Res*. 2018; 143: 503-517.
114. González-Fernández C, Méndez L, Tomas-Pejó E, Ballesteros M. Biogas and volatile fatty acids production: Temperature as a determining factor in the anaerobic digestion of *Spirulina platensis*. *Waste Biomass Valoriz*. 2019; 10: 2507-2515.
115. Shen R, Jing Y, Feng J, Zhao L, Yao Z, Yu J, et al. Simultaneous carbon dioxide reduction and enhancement of methane production in biogas via anaerobic digestion of cornstalk in continuous stirred-tank reactors: The influences of biochar, environmental parameters, and microorganisms. *Bioresour Technol*. 2021; 319: 124146.
116. Ünyay H, Yılmaz F, Başar İA, Perendeci NA, Çoban I, Şahinkaya E. Effects of organic loading rate on methane production from switchgrass in batch and semi-continuous stirred tank reactor system. *Biomass Bioenergy*. 2022; 156: 106306.
117. Yu Q, Sun C, Liu R, Yellezuome D, Zhu X, Bai R, et al. Anaerobic co-digestion of corn stover and chicken manure using continuous stirred tank reactor: The effect of biochar addition and urea pretreatment. *Bioresour Technol*. 2021; 319: 124197.
118. Kress P, Nägele HJ, Oechsner H, Ruile S. Effect of agitation time on nutrient distribution in full-scale CSTR biogas digesters. *Bioresour Technol*. 2018; 247: 1-6.
119. He H, Liu L, Ma H. The key regulative parameters in pilot-scale IC reactor for effective incineration landfill leachate treatment: Focus on the process performance and microbial community. *J Water Process Eng*. 2023; 51: 103322.
120. Li H, Wang C, Chen X, Xiong L, Guo H, Yao S, et al. Anaerobic digestion of rice straw pretreatment liquor without detoxification for continuous biogas production using a 100 L internal circulation reactor. *J Clean Prod*. 2022; 349: 131450.
121. Hellal MS, El-Kamah HM, Doma HS. High-performance internal circulation anaerobic granular sludge reactor for cattle slaughterhouse wastewater treatment and simultaneous biogas production. *BMC Biotechnol*. 2024; 24: 29.
122. Huang Y, Ma Y, Wan J, Wang Y. Mathematical modelling of the internal circulation anaerobic reactor by anaerobic digestion model No. 1, simultaneously combined with hydrodynamics. *Sci Rep*. 2019; 9: 6249.
123. Feldman H, Flores-Alsina X, Kjellberg K, Jeppsson U, Batstone DJ, Gernaey KV. Model-based analysis and optimization of a full-scale industrial high-rate anaerobic bioreactor. *Biotechnol Bioeng*. 2018; 115: 2726-2739.
124. Hu Y, Cai X, Du R, Yang Y, Rong C, Qin Y, et al. A review on anaerobic membrane bioreactors for enhanced valorization of urban organic wastes: Achievements, limitations, energy balance and future perspectives. *Sci Total Environ*. 2022; 820: 153284.
125. Mahmood Z, Cheng H, Tian M. A critical review on advanced anaerobic membrane bioreactors (AnMBRs) for wastewater treatment: Advanced membrane materials and energy demand. *Environ Sci Water Res Technol*. 2022; 8: 2126-2144.

126. Jiang M, Westerholm M, Qiao W, Wandera SM, Dong R. High rate anaerobic digestion of swine wastewater in an anaerobic membrane bioreactor. *Energy*. 2020; 193: 116783.
127. Robles Á, Jiménez-Benítez A, Giménez JB, Durán F, Ribes J, Serralta J, et al. A semi-industrial scale AnMBR for municipal wastewater treatment at ambient temperature: Performance of the biological process. *Water Res*. 2022; 215: 118249.
128. Harris PW, McCabe BK. Process optimisation of anaerobic digestion treating high-strength wastewater in the Australian red meat processing industry. *Appl Sci*. 2020; 10: 7947.
129. de Paula Sousa I, Rosa AP, Lopes JO, dos Reis Magos B, Cecon PR, Perez R, et al. Study of internal and external temperatures and their influence on covered lagoon digester performance. *Biomass Bioenergy*. 2022; 159: 106380.
130. Galván-Arzola U, Valencia-Vázquez R, Gómez-González R, Alcalá-Rodríguez MM, Loredó-Medrano JÁ, García-Balandrán EE, et al. Low-performance diagnosis of covered anaerobic lagoons as a waste management strategy in the intensive dairy industry. *Environ Technol*. 2025; 46: 772-784.
131. Lovanh N, Loughrin J, Ruiz-Aguilar G, Sistani K. Methane production from a rendering waste covered anaerobic digester: Greenhouse gas reduction and energy production. *Energies*. 2023; 16: 7844.
132. Atelge MR, Senol H, Djaafri M, Hansu TA, Krisa D, Atabani A, et al. A critical overview of the state-of-the-art methods for biogas purification and utilization processes. *Sustainability*. 2021; 13: 11515.
133. Li Y, Alaimo CP, Kim M, Kado NY, Peppers J, Xue J, et al. Composition and toxicity of biogas produced from different feedstocks in California. *Environ Sci Technol*. 2019; 53: 11569-11579.
134. Fu S, Angelidaki I, Zhang Y. *In situ* biogas upgrading by CO₂-to-CH₄ bioconversion. *Trends Biotechnol*. 2021; 39: 336-347.
135. Zhao J, Li Y, Dong R. Recent progress towards in-situ biogas upgrading technologies. *Sci Total Environ*. 2021; 800: 149667.
136. Khan MU, Lee JT, Bashir MA, Dissanayake PD, Ok YS, Tong YW, et al. Current status of biogas upgrading for direct biomethane use: A review. *Renew Sustain Energy Rev*. 2021; 149: 111343.
137. Sarker S, Lamb JJ, Hjelm DR, Lien KM. Overview of recent progress towards *in-situ* biogas upgradation techniques. *Fuel*. 2018; 226: 686-697.
138. Angelidaki I, Treu L, Tsapekos P, Luo G, Campanaro S, Wenzel H, et al. Biogas upgrading and utilization: Current status and perspectives. *Biotechnol Adv*. 2018; 36: 452-466.
139. Shonhiwa C, Mapantsela Y, Makaka G, Mukumba P, Shambira N. Biogas valorisation to biomethane for commercialisation in South Africa: A review. *Energies*. 2023; 16: 5272.
140. Awe OW, Zhao Y, Nzihou A, Minh DP, Lyczko N. A review of biogas utilisation, purification and upgrading technologies. *Waste Biomass Valoriz*. 2017; 8: 267-283.
141. Ahmed SF, Mofijur M, Tarannum K, Chowdhury AT, Rafa N, Nuzhat S, et al. Biogas upgrading, economy and utilization: A review. *Environ Chem Lett*. 2021; 19: 4137-4164.
142. Sulewski P, Ignaciuk W, Szymańska M, Wąs A. Development of the biomethane market in Europe. *Energies*. 2023; 16: 2001.
143. Khan IU, Othman MH, Hashim H, Matsuura T, Ismail AF, Rezaei-DashtArzhandi M, et al. Biogas as a renewable energy fuel-A review of biogas upgrading, utilisation and storage. *Energy Convers Manage*. 2017; 150: 277-294.

144. Struk M, Kushkevych I, Vítězová M. Biogas upgrading methods: Recent advancements and emerging technologies. *Rev Environ Sci Bio Technol*. 2020; 19: 651-671.
145. Tratz P, Torre M, Paolini V, Tomassetti L, Montiroli C, Manzo E, et al. Liquefied biomethane for heavy-duty transport in Italy: A well-to-wheels approach. *Transp Res D Transp Environ*. 2022; 107: 103288.
146. Daniel-Gromke J, Rensberg N, Denysenko V, Stinner W, Schmalfuß T, Scheftelowitz M, et al. Current developments in production and utilization of biogas and biomethane in Germany. *Chem Ing Tech*. 2018; 90: 17-35.
147. IRENA. Renewable Energy Statistics 2023 [Internet]. Abu Dhabi, United Arab Emirates: IRENA; 2023. Available from: https://www.irena.org/-/media/Files/IRENA/Agency/Publication/2023/Jul/IRENA_Renewable_energy_statistics_2023.pdf.
148. Prussi M, Padella M, Conton M, Postma ED, Lonza L. Review of technologies for biomethane production and assessment of Eu transport share in 2030. *J Clean Prod*. 2019; 222: 565-572.
149. de Carvalho JC, de Souza Vandenberghe LP, Sydney EB, Karp SG, Magalhães Jr AI, Martinez-Burgos WJ, et al. Biomethane production from sugarcane vinasse in a circular economy: Developments and innovations. *Fermentation*. 2023; 9: 349.
150. Zheng L, Cheng S, Han Y, Wang M, Xiang Y, Guo J, et al. Bio-natural gas industry in China: Current status and development. *Renew Sustain Energy Rev*. 2020; 128: 109925.
151. Xue S, Song J, Wang X, Shang Z, Sheng C, Li C, et al. A systematic comparison of biogas development and related policies between China and Europe and corresponding insights. *Renew Sustain Energy Rev*. 2020; 117: 109474.
152. Dumont KB, Hildebrandt D, Sempuga BC. The “yuck factor” of biogas technology: Naturalness concerns, social acceptance and community dynamics in South Africa. *Energy Res Soc Sci*. 2021; 71: 101846.
153. Department of Climate Change, Energy, the Environment and Water. Australian Energy Update 2024 [Internet]. Canberra, Australia: Department of Climate Change, Energy, the Environment and Water; 2024. Available from: <https://www.energy.gov.au/publications/australian-energy-update-2024>.
154. Singh N, Singhanian RR, Nigam PS, Dong CD, Patel AK, Puri M. Global status of lignocellulosic biorefinery: Challenges and perspectives. *Bioresour Technol*. 2022; 344: 126415.
155. D’adamio I, Sassanelli C. Biomethane community: A research agenda towards sustainability. *Sustainability*. 2022; 14: 4735.
156. Banu JR, Kavitha S, Tyagi VK, Gunasekaran M, Karthikeyan OP, Kumar G. Lignocellulosic biomass based biorefinery: A successful platform towards circular bioeconomy. *Fuel*. 2021; 302: 121086.
157. Lu H, Yadav V, Bilal M, Iqbal HM. Bioprospecting microbial hosts to valorize lignocellulose biomass-environmental perspectives and value-added bioproducts. *Chemosphere*. 2022; 288: 132574.
158. De Bhowmick G, Sarmah AK, Sen R. Lignocellulosic biorefinery as a model for sustainable development of biofuels and value added products. *Bioresour Technol*. 2018; 247: 1144-1154.
159. Zhao X, Joseph B, Kuhn J, Ozcan S. Biogas reforming to syngas: A review. *iScience*. 2020; 23: 101082.

160. Zain MM, Mohamed AR. An overview on conversion technologies to produce value added products from CH₄ and CO₂ as major biogas constituents. *Renew Sustain Energy Rev.* 2018; 98: 56-63.
161. Assunção LR, Mendes PA, Matos S, Borschiver S. Technology roadmap of renewable natural gas: Identifying trends for research and development to improve biogas upgrading technology management. *Appl Energy.* 2021; 292: 116849.
162. Bagi Z, Ács N, Böjti T, Kakuk B, Rákhely G, Strang O, et al. Biomethane: The energy storage, platform chemical and greenhouse gas mitigation target. *Anaerobe.* 2017; 46: 13-22.
163. Baena-Moreno FM, Sebastia-Saez D, Wang Q, Reina TR. Is the production of biofuels and biochemicals always profitable? Co-production of biomethane and urea from biogas as case study. *Energy Convers Manage.* 2020; 220: 113058.
164. Pinales-Márquez CD, Rodríguez-Jasso RM, Araújo RG, Loredó-Trevino A, Nabarlatz D, Gullón B, et al. Circular bioeconomy and integrated biorefinery in the production of xylooligosaccharides from lignocellulosic biomass: A review. *Ind Crops Prod.* 2021; 162: 113274.
165. Dale BE, Bozzetto S, Couturier C, Fabbri C, Hilbert JA, Ong R, et al. The potential for expanding sustainable biogas production and some possible impacts in specific countries. *Biofuel Bioprod Biorefin.* 2020; 14: 1335-1347.
166. Alhawari O, Awan U, Bhutta MK, Ülkü MA. Insights from circular economy literature: A review of extant definitions and unravelling paths to future research. *Sustainability.* 2021; 13: 859.
167. Dahmen N, Lewandowski I, Zibek S, Weidtmann A. Integrated lignocellulosic value chains in a growing bioeconomy: Status quo and perspectives. *Glob Change Biol Bioenergy.* 2019; 11: 107-117.
168. Ali S, Dar MA, Liaqat F, Sethupathy S, Rani A, Khan MI, et al. Optimization of biomethane production from lignocellulosic biomass by a developed microbial consortium. *Process Saf Environ Prot.* 2024; 184: 1106-1118.
169. Catalano G, D'Adamo I, Gastaldi M, Nizami AS, Ribichini M. Incentive policies in biomethane production toward circular economy. *Renew Sustain Energy Rev.* 2024; 202: 114710.
170. D'Adamo I, Ribichini M, Tsagarakis KP. Biomethane as an energy resource for achieving sustainable production: Economic assessments and policy implications. *Sustain Prod Consum.* 2023; 35: 13-27.
171. Solarte-Toro JC, Rueda-Duran CA, Ortiz-Sanchez M, Alzate CA. A comprehensive review on the economic assessment of biorefineries: The first step towards sustainable biomass conversion. *Bioresour Technol Rep.* 2021; 15: 100776.
172. Bose A, O'Shea R, Lin R, Long A, Rajendran K, Wall D, et al. The marginal abatement cost of co-producing biomethane, food and biofertiliser in a circular economy system. *Renew Sustain Energy Rev.* 2022; 169: 112946.
173. Junior AD, Etchebehere C, Perecin D, Teixeira S, Woods J. Advancing anaerobic digestion of sugarcane vinasse: Current development, struggles and future trends on production and end-uses of biogas in Brazil. *Renew Sustain Energy Rev.* 2022; 157: 112045.
174. Wang Y, Fang J, Lü F, Zhang H, He P. Food waste anaerobic digestion plants: Underestimated air pollutants and control strategy. *Sci Total Environ.* 2023; 903: 166143.
175. Paolini V, Petracchini F, Segreto M, Tomassetti L, Naja N, Cecinato A. Environmental impact of biogas: A short review of current knowledge. *J Environ Sci Health A.* 2018; 53: 899-906.

176. Malet N, Pellerin S, Girault R, Nesme T. Does anaerobic digestion really help to reduce greenhouse gas emissions? A nuanced case study based on 30 cogeneration plants in France. *J Clean Prod.* 2023; 384: 135578.
177. Thompson TM, Ramin P, Udugama I, Young BR, Gernaey KV, Baroutian S. Techno-economic and environmental impact assessment of biogas production and fertiliser recovery from pelagic sargassum: A biorefinery concept for Barbados. *Energy Convers Manage.* 2021; 245: 114605.
178. van den Oever AE, Cardellini G, Sels BF, Messagie M. Life cycle environmental impacts of compressed biogas production through anaerobic digestion of manure and municipal organic waste. *J Clean Prod.* 2021; 306: 127156.
179. Karthikeyan PK, Bandulasena HC, Radu T. A comparative analysis of pre-treatment technologies for enhanced biogas production from anaerobic digestion of lignocellulosic waste. *Ind Crops Prod.* 2024; 215: 118591.
180. Chiaraluce G, Bentivoglio D, Finco A. Circular economy for a sustainable agri-food supply chain: A review for current trends and future pathways. *Sustainability.* 2021; 13: 9294.
181. D'Adamo I, Falcone PM, Huisingsh D, Morone P. A circular economy model based on biomethane: What are the opportunities for the municipality of Rome and beyond? *Renewable Energy.* 2021; 163: 1660-1672.
182. Wu B, Lin R, Bose A, Huerta JD, Kang X, Deng C, et al. Economic and environmental viability of biofuel production from organic wastes: A pathway towards competitive carbon neutrality. *Energy.* 2023; 285: 129322.
183. Scandurra F, Salomone R, Caeiro S, Gulotta TM. The maturity level of the agri-food sector in the circular economy domain: A systematic literature review. *Environ Impact Assess Rev.* 2023; 100: 107079.