Renewable and Sustainable Energy Reviews 170 (2022) 112995

Contents lists available at ScienceDirect

Renewable and Sustainable Energy Reviews

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

Advances in biological techniques for sustainable lignocellulosic waste utilization in biogas production

Zhenghui Gao [a], Khaled Alshehri [a,b], Yuan Li [c,a], Hang Qian [a], Devin Sapsford [a], Peter Cleall [a], Michael Harbottle [a,*]

[a] School of Engineering, Cardiff University, Cardiff CF24 3AA, UK
[b] Civil Engineering Department, College of Engineering, University of Bisha, Bisha, 67714, Saudi Arabia
[c] State Key Laboratory of Geomechanics and Geotechnical Engineering, Institute of Rock and Soil Mechanics, Chinese Academy of Sciences, Wuhan, 430071, China

ARTICLE INFO

Keywords:
Lignocellulosic waste
Anaerobic digestion
Biogas production
Biomethane
Biological techniques

ABSTRACT

Improper disposal of lignocellulosic wastes may produce a large quantity of greenhouse gases and pollute the environment. Through anaerobic digestion processes, lignocellulosic wastes can be recycled to produce clean and renewable biogas. However, the lignin in lignocellulose limits its potential as such a biomass resource, and the efficacy of biogas production is not satisfactory although recent research efforts have attempted to address this issue. In this review, the physicochemical characteristics of three lignocellulosic wastes, including municipal solid waste, forestry waste, and crop straw, are summarized. Then, the mechanism and influencing factors of biogas production from these wastes through anaerobic digestion are presented. Biological pretreatment techniques have been confirmed to increase lignocellulose hydrolysis and then enhance biogas production, among them, co-culture systems, metabolic engineering and anaerobic co-digestion are worthy of focus in future research. Furthermore, natural lignocellulose degrading systems, like xylophagous insects and ruminants, also have potential for improving the anaerobic digestion system. This review also considers the future perspective of anaerobic digestion of lignocellulosic wastes, including kinetics and model studies to optimize anaerobic digestion process, and policy to facilitate biogas production from lignocellulosic wastes. This article aims to comprehensively review challenges with anaerobic digestion of lignocellulosic wastes and summarize available pretreatment methods focusing mainly on biological techniques to find efficient and low-cost strategies for improving the anaerobic digestion process and biogas production.

1. Introduction

It is reported that fossil fuels comprise approximately 80% of current global primary energy usage, and fossil fuel based energy systems are primarily responsible for more than two thirds of global greenhouse gas (GHG) emissions [1]. Therefore, developing alternative energies through technological innovation becomes a priority. Anaerobic digestion of biomass into biogas has been given much attention in recent years. Biogas production in 2018 was around 35 million tons of oil equivalent (Mtoe), only a fraction of the estimated overall potential (730 Mtoe) [2]. It is estimated that the availability of sustainable feedstocks for producing biogas is set to grow by 40% over the period to 2040, which avoids around 1000 million tons of GHG emissions [2].

Lignocellulosic waste materials are available in significant quantities but presently only contribute a limited fraction of biogas output. The most available lignocellulosic waste materials are agricultural by-products (e.g. crop stalks and straws), wood and branches from forest management and commercial timber production and some fractions of

Abbreviations: IEA, International Energy Agency; EPA, Environmental Protection Agency; FAOSTAT, Food and Agriculture Organization Corporate Statistical Database; Mtoe, Million tons of oil equivalent; GHG, Global greenhouse gas; WD, Woody debris; MSW, Municipal solid waste; GMO, Genetically modified organism; AD, Anaerobic digestion; CSTR, A continuous flow stirred tank reactor; VFAs, Volatile fatty acids; VS, Volatile solids; PS, Particle size; L/(C + H), Lignin content/(Cellulose content + Hemicellulose content); LCH, Lignin content + Cellulose content + Hemicellulose content; I/S, Inoculum to substrate; TMP, Theoretical methane potential; COD, Chemical oxygen demand; BMP, Biochemical methane potential; SW, Softwood; HW, Hardwood; OSB, Oriented strand board; MDF, Medium density fiberboard; AcoD, Anaerobic co-digestion.

* Corresponding author.
E-mail address: harbottlem@cardiff.ac.uk (M. Harbottle).

https://doi.org/10.1016/j.rser.2022.112995
Received 21 June 2022; Received in revised form 7 October 2022; Accepted 13 October 2022
Available online 21 October 2022
1364-0321/

municipal solid waste (e.g. wood, paper and paperboard) [3], which at present are not recovered for biogas generation in significant quantities and could emit GHG if being left to degrade where they are produced or transported to landfill [4,5]. However, the main hurdles in utilizing lignocellulosic waste are presented by the lack of biodegradability of lignin, a limited accessible surface area for enzymatic hydrolysis, and cellulose crystallinity [6–8]. Efficient delignification and improved digestibility of cellulose and hemicellulose in lignocellulosic waste is usually a crucial step of pretreatment. In this respect, biological pretreatment is an attractive pretreatment technology with several significant advantages, including simple operating conditions and equipment, low energy consumption, no or minimum inhibitor formation, and no requirement to remove solvents after pretreatment [9,10]. It has already been widely applied in lab-scale bioreactors or even full-scale biorefinery plants and is gaining in popularity [11,12]. Despite the huge potential, large-scale implementation of biological pretreatment is still constrained by issues such as long pretreatment times, loss of carbohydrates, and low downstream yields [13].

To achieve the technoeconomic feasibility for the large-scale exploitation of biological pretreatment, substantial research efforts remain essential. This review therefore aims to provide an in-depth and critical appraisal of the status of biological pretreatment of lignocellulosic wastes under the following structure: (1) summary of types and content of lignocellulosic waste (Section 2); (2) overview on biogas production from anaerobic digestion of lignocellulosic waste (Section 3), including process mechanisms, major factors affecting biogas production, lignocellulosic waste recalcitrance, and importance of different lignocellulose and fermentation parameters; (3) review of available biotechnology pretreatments for lignocellulosic wastes (Section 4); (4) consideration of economic aspects of biogas production from lignocellulosic waste (Section 5); (5) discussion of challenges, limitations, and perspectives (Section 6).

2. Types of lignocellulosic waste utilized for biogas production

Lignocellulosic biomass comprises dry plant materials and so covers many substances including different grasses, plant stems, trees, and residues from modern sawmills and paper mills. It can be broadly classified into virgin biomass, energy crops and waste biomass. Nearly 200 billion tons of lignocellulosic waste are generated globally every year [14], mostly low value byproduct from various industrial sectors, human activities or the natural environment such as agriculture (e.g. crop straw and stalk), municipal waste (e.g. wood, paper and cardboard) and forestry (e.g. sawmills and paper mills discards, forest management waste). All of these lignocellulosic wastes have been recognized as valuable resources by the U.S. Department of Energy (DOE) [15].

2.1. Municipal solid waste

MSW mainly comprises of commercial wastes, residential wastes and yard wastes generated in municipal areas in either semisolid or solid form excluding agricultural wastes and industrial hazardous wastes but including treated biomedical wastes [16]. The global level of MSW generation is estimated to increase to approximately 2.2 billion tons by 2025 [17]. Low- and middle-income regions produce most MSW, accounting for up to 90.4% of the total [17]. The yearly MSW generation in America and China are highest in the world at 292.36 million tons (2.49 kg per capita per day) and 228.02 million tons (0.45 kg per capita per day), respectively (Fig. 1A) [18,19]. The disposal of MSW depends on national development levels. Landfilling and thermal treatment are valued in high income countries, while composting, and open dumping still account for a large proportion in low- and lower-middle income countries [17]. Fig. 1B shows the data from the Organisation for Economic Co-operation and Development (OECD) [20], U.S. Environmental Protection Agency (EPA) [21] and Eurostat [22], with different countries having very different combinations of waste disposal methods based on their own development and national conditions, however globally sanitary landfilling is currently the dominant MSW disposal method.

In most developing countries, MSW is not segregated at source and is transported into landfill in mixed conditions [23]. An average composition of MSW in different countries is presented in Table 1, showing that there exists a large percentage of lignocellulosic materials and suitable feedstocks for biogas, i.e., paper and paperboard, wood and yard waste. Of these, wood, paper and paperboard wastes are likely to have more recycled value, since they can be reprocessed into particleboard or new cardboard [24]. However, a significant fraction of wood and paper wastes are nonrecyclable with chemical preservatives, binders or metal protectants [25]. The main destination of these lignocellulosic wastes is landfill, Table 2 shows lignocellulosic waste data from 1990 to 2018 in the USA [21]. About 15.6 million tons of wood waste is produced each year, with 71.5% landfilled. An average of 74.2 million tons of paper and cardboard wastes are produced each year, with 36.9% landfilled.

Landfilling, if inappropriately performed or poorly operated, may contaminate the atmosphere with GHG emissions from the slow degradation of lignocellulosic waste. The presence of abundant lignin endows lignocellulosic waste with recalcitrance; it is difficult to break down through microbial action [31]. These poorly degradable fractions are typically associated with a long 'tail' of emissions and gradually accumulate in landfill. Furthermore, these slowly produced gases are insufficient to generate energy and difficult to capture, so the biogas (primarily methane and carbon dioxide) typically escapes into the atmosphere contributing to climate change [32].

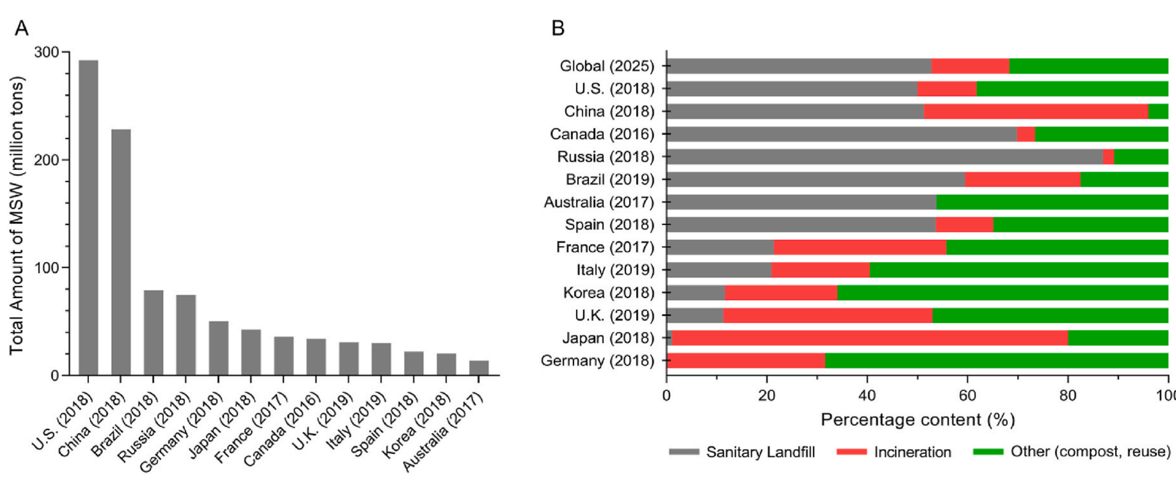


Fig. 1. The total generation amount (A) and disposal methods (B) of MSW in different countries [18–22].

Table 1
National average municipal solid waste composition.

Country	Organic (%)	Paper (%)	Plastic (%)	Glass (%)	Metal (%)	Other (%)
Global	46	17	10	5	4	18
Low Income	62	6	9	3	3	17
Lower Middle Income	55	10	13	4	3	15
Upper Middle Income	50	15	12	4	4	15
High Income	28	30	11	7	6	18
China	59	8	10	3	1	19
India	40	10	2	0.2	0	47.8
U.S.	39.9	23.1	12.2	4.2	8.8	11.8
Russia	40	19	14	12	4	11
Brazil	51.4	13.1	13.5	2.4	2.9	16.7
Indonesia	74	10	8	2	2	4
Nigeria	68	10	7	4	3	8
Pakistan	67	5	18	2	0	7
U.K.	46	17	10	7	5	15
Germany	14	34	22	12	5	12
Netherlands	35	26	19	4	4	12
Australia	47	23	4	7	5	13
Mexico	51	15	6	6	3	18
Portugal	34	21	11	7	4	23
Italy	44.5	19.1	8.3	12.3	2	13.8
Spain	44	18	13	9	4	12
Japan	34	34	11.8	4.3	4.7	11.2
Canada	47	15	13	2	3	20

Note: Classification according to Daniel and Perinaz [17]. Organic: food scraps, yard waste, wood, process residues; Paper: paper scraps, cardboard, newspapers, magazines, bags, boxes, wrapping paper, telephone books, shredded paper, paper beverage cups; Plastic: bottles, packaging, containers, bags, lids, cups; Glass: bottles, broken glassware, light bulbs, colored glass; Metal: cans, foil, tins, non-hazardous aerosol cans, appliances (white goods), railings, bicycles; Other: textiles, leather, rubber, multi-laminates, e-waste, appliances, ash, other inert materials. All data are adapted from OECD [20], EPA [21], Statista [26], Ding et al. [27], Khan et al. [28], Millati et al. [29] and Alfaia et al. [30].

Accelerating the degradation of lignin and subsequent methanogenesis in lignocellulosic waste is required to help to confine methane production to a shorter period of higher concentration release, thus allowing more landfill biogas to be collected as energy and preventing low emission of GHG in the long term. This can be addressed with biotechnological methods in two main ways – the application of extracellular enzymes [33–35] or enzyme-producing microorganisms [36–39]. However, these previously mentioned studies have been carried out under laboratory level with highly controlled conditions or standardized materials, scaling-up these technologies to landfill conditions will be highly challenging and is yet to attract significant attention.

2.2. Forestry waste

Forests are the largest terrestrial carbon sink and play a vital role in the global carbon cycle, where plants absorb energy through photosynthesis and store it in wood as carbon [40,41]. The current carbon stock in forests is estimated to be 861 billion tons, with the vast majority in soil (44%) and live biomass (42%) [42]. There are two main types of wood, hardwood and softwood. Hardwoods arise from deciduous trees (e.g. oak, maple, birch) while coniferous trees (e.g. pine, spruce, juniper) produce softwoods. In 2020, global hardwood lumber production reached 2536.7 million m^3, mainly in Asia, Africa, the Americas and Europe (Table 3) [43]. Global softwood production was 1375.25 million m^3, mostly in Europe and the Americas.

Woody debris (WD), comprising fallen dead trees and the remains of large branches the ground in forests, represents a large carbon pool with carbon stock ranging from 36 to 72 billion tons globally [44]. Unless WD is harvested it will ultimately convert to lignoforms (humus forms formed by the degradation of deadwood) as a part of the soil [45]. In this process of WD being gradually decayed into lignoforms by decomposer communities, most of the carbon is returned to the atmosphere as methane and carbon dioxide [4]. Since methane and carbon dioxide are both greenhouse gases of great concern for climate change [46], this natural process was only recently recognized as an important sources of GHG, with estimates of carbon flux at 8.6 billion tons annually, equivalent to approximately 90% of anthropogenic emissions [47].

Excepting dead wood produced by natural processes, human activities are the main source of WD. The harvesting of approximately 4.3 billion m^3 of wood annually [48] is estimated to generate 232.94 million m^3 of wood waste may be produced in the world every year, mainly in Asia and Europe (Table 3). WD are generated during forestry operations (branches, treetop, leaves, stumps, low grade and decayed wood, slashings, sawdust) and wood processing (bark, sawdust, trimmings, planer shavings, core, screening fines), which also are classed as wood waste. These wood wastes are potential resource for bioenergy

Table 3
Quantity of hardwood and softwood production and residues in 2020.

Types	Production quantity (million m^3)	Waste quantity (million m^3)	Regions
Hardwoods	995.08	102.24	Asia
	760.64	1.56	Africa
	515.78	23.49	Americas
	58.77	0.59	Central America
	134.05	5.25	Northern America
	317.33	20.05	South America
	231.87	17.57	Europe
	33.34	0.99	Oceania
	2536.7	151.05	World
Softwoods	163.83	16.83	Asia
	30.97	0.06	Africa
	555.14	25.28	Americas
	35.64	0.35	Central America
	427.83	16.77	Northern America
	91.17	5.76	South America
	571.81	43.33	Europe
	53.5	1.59	Oceania
	1375.25	81.89	World

Table 2
1960–2018 Data on MSW lignocellulosic waste in United States (ten thousand tons).

Types	Management Pathway	1960	1970	1980	1990	2000	2005	2010	2015	2017	2018
Wood	Generated	303	372	701	1221	1357	1479	1571	1630	1820	1809
	Recycled	–	–	–	13	137	183	228	266	303	310
	Incinerate	–	1	15	208	229	227	231	257	288	284
	Landfill	303	371	686	1000	991	1069	1112	1107	1229	1215
Paper and paperboard	Generated	2999	4431	5516	7273	8774	8484	7131	6805	6701	6739
	Recycled	508	677	1174	2023	3756	4196	4457	4532	4417	4597
	Incinerate	–	15	86	893	973	780	474	445	449	420
	Landfill	2491	3739	4256	4357	4045	3508	2200	1828	1835	1722

Note: means no data. Data is adapted from US EPA [21].

production that may have a significant impact on the profitability of the entire timber trade value chain, offsetting the negative impacts of forestry operations on ecosystem services and biodiversity. The added value of producing biofuel from these wood waste also comes from reducing fire risk, mitigating forest management costs, and eliminating additional emissions from degradation [49,50].

Direct large-scale combustion of wood waste to generate energy or electricity is no longer considered an efficient and environmentally appropriate option. Thus, attention must be paid to develop alternative options for renewable gaseous biofuel [51]. However, due to the refractory nature of wood waste and immaturity at application level of current anaerobic digestion technology, there are few practical examples of using wood waste fermentation to produce gaseous biofuel. The lignin content in wood is quite high, 25–39% in softwood and 18–25% in hardwood respectively [29], which is not conducive to the degradation process of microorganisms or enzymes. To make fuller use of wood waste to generate more biogas, researchers screen for new high-efficiency lignin-degrading microorganisms [52,53], or use the addition of other nitrogen-rich wastes, such as food waste or animal manure, to create a favorable condition for fermentation [54].

2.3. *Crop straw*

Agriculture wastes mainly include crop residues and livestock excreta, among which crop straw is a potentially valuable lignocellulosic waste with huge yields. As a by-product of grain production, crop straw is inevitable and its corresponding relationship with grain output is shown in Table 4 [55,56]. Based on the Food and Agriculture Organization Corporate Statistical Database (FAOSTAT) [57], the average annual crop straw production in the world from 2010 to 2022 can be calculated (Table 4). The amount of sugarcane bagasse ranked top in the world at 18575.1 million tons, followed with rice straw, corn straw, wheat straw and barley straw respectively, and the last are cotton and fiber crops. Asia is the region that produces the most food with East Asian countries like China and India major growers of crops [58]. It is estimated that 1000 million tons of crop straw are produced yearly in China [59], while India produces a total of 500 million tons [60].

Crop straw has a low nutritional value and so only a limited amount has been traditionally used as livestock feed with the rest commonly burned in the field or sent to landfills [61]. Open burning of crop straw not only produces particulate matter posing a serial health risk but also is a major cause of environmental pollution, including greenhouse gases and soil fertility destruction [62,63]. Crop straw burning varies by different countries, depending on the type of crop straw and the pattern of its management. Chen et al. claimed that Chinese farmers burned approximately 25% of crop straw [64], while this ratio would rise to 50% in line with FAOSTAT [57]. China, India, United States, Brazil, Russian Federation, Indonesia, Argentina, Nigeria, Ukraine and Thailand are the top 10 countries in terms of quantity of burned crop straw (Table 5). The burning of crop straw leads to inefficient utilization of agricultural waste and an increase in air pollution, which has drawn attention in various parts of the world to develop a proper plan for managing crop straw. Over the past few years, especially since 2015, different international agencies have proposed many avenues to utilize crop straw to minimize crop straw related issues [65].

A core sustainable development goal is the transition to a circular economy, which involves minimizing resource inputs and waste outputs within a closed-loop system pioneering wastes as secondary resources [66,67]. Use of crop straw as material to generate biogas through anaerobic digestion is in line with achieving a circular economy. The biogas production rate of main crop straw residues is shown in Table 6 [55,56]. As a clean renewable energy, biogas can alleviate energy shortages and minimize air pollution risk from the improper management of crop straw. There is little biogas production from agricultural waste currently, although the supply of raw crop straw is plentiful. In India, only 2.07 billion m^3 biogas are currently produced per year, though there is the potential for 29–48 billion m^3 each year based on straw volume [68]. The biogas industry of China is considered to have great potential, owing to tremendous amount of crop straw. Nevertheless, the ratio of actual biogas production to total biogas potential is only 6.17% [69]. The biogas potential of crop straw is still underexplored due to an imperfect supply chain and viable business models, lack of simple pre-treatment technologies, insufficient short-term returns, and shortage of advanced technology [60]. Many small-scale biogas plants have been operating for decades, although large-scale technically advanced biogas plants are uncommon and a recent development [70]. The priority currently is to improve the biogas potential from crop straw, which could help to eliminate air pollution threats and develop clean energy.

Table 5
Top 10 countries of crop straw burning in the world in 2019.

Countries	Biomass burned (million tons)	CH_4 emission (kilotons)	N_2O emission (kilotons)
China	68.2	184.2	4.8
India	48.1	129.9	3.4
USA	39.8	107.3	2.8
Brazil	25.9	69.8	1.8
Russian	13.6	36.8	1.0
Indonesia	11.8	31.9	0.8
Argentina	10.1	27.2	0.7
Nigeria	9.8	26.6	0.7
Ukraine	7.7	20.9	0.5
Thailand	7.5	20.2	0.5

Table 4
Quantities of crop straw reportedly by region, average 2010–2020 (million tons).

Types	Ratio of Straw/Grain	Crop production						Straw production					
		World	Asia	Americas	Europe	Africa	Oceania	World	Asia	Americas	Europe	Africa	Oceania
Rice	1.6	734.8	661.7	36.9	4.2	31.4	0.6	1175.7	1058.7	59.0	6.7	50.2	1.0
Barley	1.0	143.3	21.4	18.7	87.1	6.5	9.5	143.3	21.4	18.7	87.1	6.5	9.5
Corn	0.5	1038.2	325.7	520.0	114.4	77.5	0.6	519.1	162.9	260.0	57.2	38.8	0.3
Wheat	0.7	724.6	321.3	113.4	240.1	25.9	23.9	507.2	224.9	79.4	168.1	18.1	16.7
Sorghum	1.3	60.7	9.0	22.1	1.1	27.0	1.6	78.9	11.7	28.7	1.4	35.1	2.1
Oat	1.3	22.9	1.1	6.2	14.1	0.2	1.3	29.8	1.4	8.1	18.3	0.3	1.7
Beans	0.7	22.0	20.1	0.3	0.9	0.7	0.044	15.4	14.1	0.2	0.6	0.5	0.031
Tubers	2.0	6.8	3.7	0.9	0.1	1.7	0.4	13.6	7.4	1.8	0.1	3.5	0.8
Cotton	0.3	24.9	16.3	5.9	0.3	1.6	0.7	7.5	4.9	1.8	0.1	0.5	0.2
Fiber crops	0.4	0.6	0.4	0.1	0	0.048	0.0042	0.2	0.2	0.1	0	0.019	0.0017
Sugarcane	10.0	1857.5	729.5	1001.8	0.0057	93.2	33.0	18575.1	7295.3	10018.4	0.1	931.9	329.5

Table 6
Dry biomass ratio and biogas production rate of crop straw.

Types of crop straw	Dry matter (%)	Carbohydrates (%)	Biogas yield (m^3/kg of dry biomass)
Rice straw	88	49.33	0.43
Barley straw	81	70.00	0.48
Corn straw	78.5	58.29	0.46
Wheat straw	90.1	54.00	0.45
Sorghum straw	88	61.00	0.41
Oat straw	89.1	59.10	0.40
Beans straw	80	54.48	0.40
Sugarcane bagasse	71	67.15	0.43

3. Biogas production from anaerobic digestion of lignocellulosic waste

3.1. Process mechanisms

Anaerobic digestion (AD) decomposes lignocellulosic feedstocks into biogas in four stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis [71] (Fig. 2). These steps are a synergistic process of diverse microbial groups, and bacterial metabolic activities at different stages mutually affect each other, in close dependence on each other.

The AD process starts with hydrolysis, at this stage hydrolytic anaerobic and facultative bacteria produce extracellular hydrolases that destroy the cellulose-hemicellulose-lignin structure and then convert complex polymeric organics to soluble monomers [71,73]. In acidogenesis, the reduced monomers (amino acids, peptides, long-chain fatty acids, glycerides, and sugars) are further degraded by facultative aerobes to volatile fatty acids (VFAs) (53–58% acetic acid, 6–13% propionic acid and 30–35% butyric acid) and other minor products (alcohols, aldehydes, hydrogen, and carbon dioxide) [74,75]. During acetogenesis, syntrophic bacteria transform previous VFAs and alcohols into acetate, carbon dioxide, and hydrogen which are substrates for the production of methane [72,74]. The final step is methanogenesis, where methanogenic archaea convert acetate, carbon dioxide, and hydrogen to generate methane, mainly consisting of acetoclastic methanogenesis (60–70%) and hydrogenotrophic methanogenesis (about 30%) [72,76].

3.2. Major factors affecting biogas production

Since all four stages take place in one bioreactor, AD is a fragile biological process that can easily be disrupted resulting in activity inefficiency or even complete failure. The equilibria of AD depend on feedstock characteristics [77,78], temperature [79,80], pH [81], microbial community [82], solid-liquid ratio [83], inhibitory compounds [84], etc. In general, temperatures of 10–65 °C [85], pH of 5.0–8.5 (5.5–7.0 for hydrolysis and acidogenesis, 6.8–8.5 for methanogenesis) [85,86], C/N ratios of 20–35 [87,88] and solids loadings of 20–40% [89] are the optimal ranges for methane yield. Temperature range can vary widely during AD, and it is divided into three categories according to the microbial activity, psychrophilic: 10–20 °C; mesophilic: 20–45 °C (usually 37 °C); and thermophilic: 50–65 °C (usually 55 °C) [85,90]. Compared to the effect of temperature, pH plays a more important role in AD. Anaerobic fermentation bacteria, especially methanogens, are sensitive to the acid concentration in the system. The growth of methanogens could be inhibited at lower pH conditions, and the optimal pH for methanogenesis has been found to be around 7.2 [91]. Improper C/N ratios will result in the release of a large amount of ammonia nitrogen or the excessive accumulation of volatile fatty acids, which are inhibitors in anaerobic digestion process [92]. Therefore, an appropriate temperature, pH and C/N ratio are needed for maintaining a stable environment in a long-term operation.

While temperature, pH, and C/N ratio can be adjusted by controlling the operating conditions, reactor configuration, and the concentrations of N-containing additives [93–95], feedstock characteristics have specific impacts on methane production. Feedstock characteristics include chemical composition [96], volatile solids (VS) content [97], chemical oxygen demand (COD) [98], morphology (particle size and porosity) [80], and nutrient content [54]. These characteristics relate to initiation rate, biodegradability of components, VS conversion rate, hydraulic retention time, and ultimately impact biochemical methane potential (BMP) [72]. In lignocellulosic waste, cellulose and hemicellulose are the major contributors to methane formation. They are easily degraded because of their less complicated molecular structure [99]. However, the presence of lignin limits methane production by reducing the surface area that bacteria can access through a rigid lignin-cellulose-hemicellulose matrix or high cellulose crystallinity, thereby suppressing biodegradability of other components [3,100]. In addition, the decomposition of lignin produces inhibitors, such as phenolic aldehydes and acids, which may also inhibit the methanogenesis process [101].

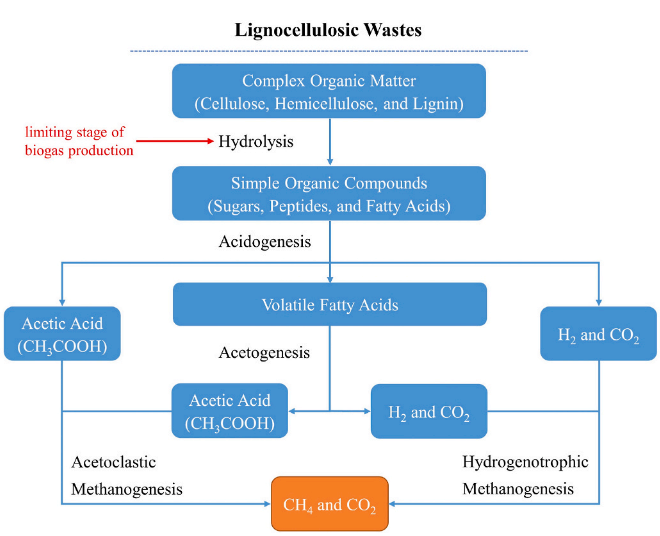


Fig. 2. Degradation pathways during anaerobic digestion of lignocellulosic waste, modified from Ref. [72].

3.3. Lignocellulosic waste recalcitrance

An extensive range of protein-rich or fat-rich wastes are the usual target for anaerobic digestion, however, less digestible lignocellulosic components are rarely exploited due to low energy extraction efficiency. Lignocellulosic biomass, formed from lignin, cellulose, and hemicellulose, presents barriers to degradation through the structure of individual components and also their combined form. Barriers arise from the interconnection between cellulose, hemicellulose, and lignin, forming a complex and undegradable lignocellulosic matrix [102].

Cellulose is an unbranched biopolymer of β-1,4 glucan, whereas hemicellulose is a heterogeneous polymer of various sugars. The glucose chains in cellulose do not exist independently and tend to produce three-dimensional microfibrils with a high degree of polymerization through van der Waals interactions and hydrogen bonds [103]. Each glucose unit is hydrogen-bonded with two intra-chains and two or three inter-chains. These hydrogen bonds give cellulose its crystallinity, which makes it structurally stable and tightly packed.

The recalcitrance of lignin has been a major obstruction for the utilization of lignocellulosic wastes. Lignin requires high temperatures and high acidity to be dissolved and considered as the most stubborn component in lignocellulose [104]. It has been widely believed that the higher the lignin content, the more recalcitrant the biomass is. Lignin is structurally composed of three hydroxycinnamyl alcohol monomers, including coniferyl, *p*-coumaryl and sinapyl alcohol, with a variety of ethers and C–C bonds [105]. Once incorporated into the lignin polymer, these substituents are distinguished by aromatic ring structures and called guaiacyl, *p*-hydroxyphenyl and syringyl substituents. Besides the content, variation in the quantity of these components has a significant impact on delignification chemistry and therefore on biomass decomposition. Guaiac lignin is reported to be more likely to C–C cross-link at C-5 position, which cannot be hydrolyzed by acids or bases, leading to their ability to prevent fiber swelling and enzyme accessibility [106]. The lignin crust has been identified as a challenge in the hydrolysis of lignocellulose because it limits the accessible surface area of polysaccharide hydrolases to substrates. Lignin crosslinks with cellulose and hemicellulose to forms a ‘glue-like’ structure (Fig. 3A), which effectively prevents microorganisms and enzymes from attacking easily degradable parts, thereby further limiting the biogas potential of lignocellulosic waste [107]. Lignin also is a source of compounds, vanillic acid and syringyl aldehyde, which could inhibit hydrolases and fermentation organisms [108]. In summary, the recalcitrance of lignocellulosic biomass is influenced by several factors, i.e., lignin barrier, cellulose crystallinity, and accessible surface area.

3.3.1. Chemical composition

Lignocellulosic waste is composed primarily of lignin, cellulose, hemicellulose (together accounting for 90% weight by dry mass), along with a small quantity of proteins and minerals [3]. Table A1 and Fig. 4 show the content of major components in different lignocellulosic wastes. The lignocellulose composition of wood or wood products, like oriented strand board, particleboard, plywood and medium density fiberboard, are similar, with a lignin content between 25% and 40% (average of about 28.8%). In contrast, the cellulose content in paper and paperboard is up to 68.6%, and crop straw is high in hemicellulose with an average value of 27.6% (Fig. 4). Both paper, paperboard and crop straw have relatively low levels of lignin (approximately 10%), and the lignin content varies widely among the different types of crop straw, ranging from a minimum content of 5.2% in maize straw to a maximum content of 26.7% of wheat straw (Table A1).

3.3.2. Linkages of different components

Cellulose is a homogeneous long-chain polymer composed of repeating D-glucose units linked by β-1,4 glycosidic bonds. These glucose monomers are present in the pyranose of a cellulose chain with six-carbon rings, and two pyranoses being connected to each other by acetal linkages [111]. Hemicellulose is a heterogeneous polysaccharide composed of arabinose, xylose, glucose, galactose, mannose and sugar acids. These monomers are bonded to each other through glycosidic and fructose ether linkages, forming a branched polymer structure [3,112]. In lignocellulose, lignin occupies the free space between cellulose and hemicellulose and crosslinks with cellulose and hemicellulose to form a rigid structure (Fig. 3A). Lignin has a helical structure formed by the polymerization of three phenylpropane monomer units, which are connected by ether and carbon-carbon linkages [113,114]. In addition, cellulose, hemicellulose, and lignin are interconnected to form numerous intrapolymer and interpolymer cross-linkages, mainly including hydrogen, ether, and ester linkages. Table 7 lists the different intrapolymer and interpolymer linkages [115], Fig. 3B shows the linkages within lignin molecule and between lignin and other components.

3.4. Performance of different lignocellulose and fermentation parameters

The chemical composition of the feedstock and fermentation parameters have a large impact on methane production, Table B1 summarizes many typical cases of biomethane production from anaerobic digestion of different lignocellulosic wastes. To explore the connection in detail, a mesophilic fermentation (about 37 °C) with the following parameters fixed, initial pH of AD (about 7), AD time (30–50 d), inoculum (sludge) was selected, which were the commonly used conditions in anaerobic digestion of lignocellulosic waste. There is a highly positive correlation between methane and biogas yield (Fig. 5A and B). Additionally, lignin content and L/(C + H) show a moderate negative correlation with methane yield, revealing that the presence of lignin could limit anaerobic digestion of lignocellulosic wastes (Fig. 5A). Wastes with single components, such as crop straws and plant residue like peel and reed, have the highest methane yield of lignocellulosic wastes at 161.2 and 191.8 mL/g VS. The methane yield of mixed woody wastes, yard wastes and leaves are 121.8 and 65.9 mL/g VS, respectively. However, co-digestion could optimize the chemical composition of feedstock and C/N ratio, finally improve methane yield (236.7 mL/g VS) (Fig. 5C).

At different C/N and I/S ratios, the fitting curves of feedstock lignocellulose composition and methane production show that cellulose and hemicellulose are positively correlated with methane production overall (Fig. 6). It is worth noting that at the recommended C/N ratio (25–30), the content of lignin is also positively correlated with methane production (Fig. 6C). Biogas yield is generally determined by lignin content and C/N ratio. Under standard temperature and pressure conditions, the methanogenesis potential of lignin (727 mL/g VS) is much higher than that of cellulose (415 mL/g VS) and hemicellulose (424 mL/g VS) [117]. In general, lignin limits methane production, but pretreatment can increase the utilization of lignin in lignocellulosic waste, for example, through the co-digestion with other nitrogen-rich wastes to achieve the best fermentation C/N ratio of materials. At the recommended C/N ratio (25–30), some studies used NaOH solution for pretreatment (Table B1). Alkali pretreatment is considered as an effective method for maximizing degradation of complex materials, in breaking ester bonds between lignin and other compounds along with preventing

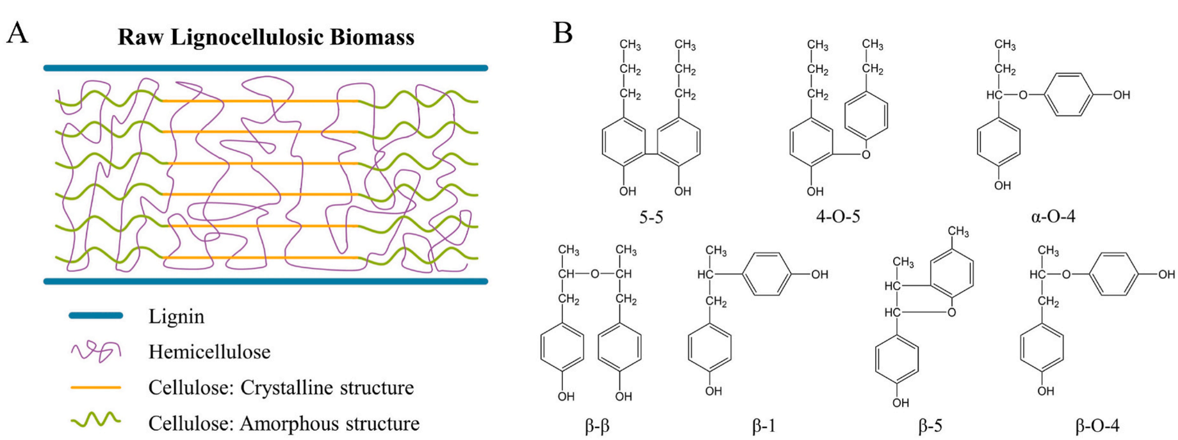


Fig. 3. (A) Spatial arrangement of lignin, cellulose, and hemicellulose in lignocellulosic biomass, modified from Ref. [7]. (B) Different types of linkages within lignin molecule and between lignin and other components, modified from Refs. [109,110].

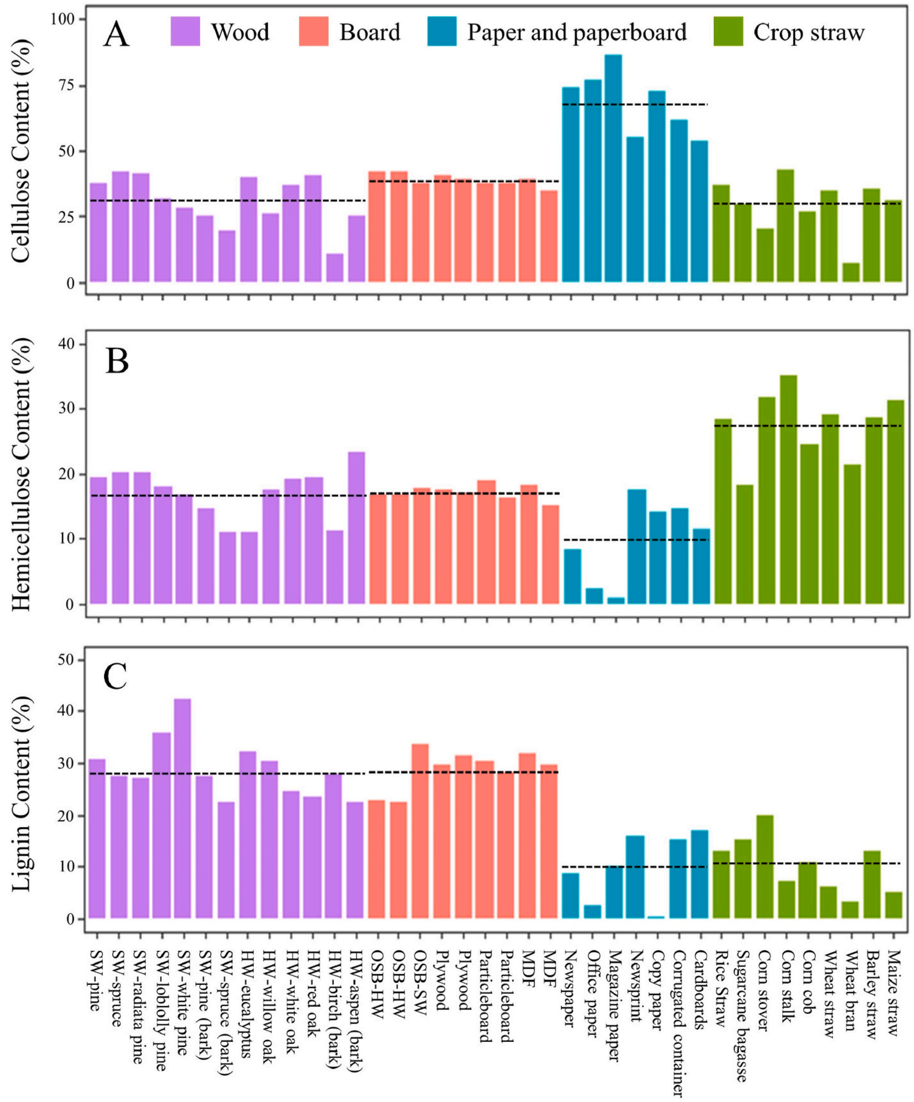


Fig. 4. The chemical composition of different lignocellulosic wastes plotted from the data in Table A1. A: Cellulose content (%); B: Hemicellulose content (%); C: Lignin content (%). Lignocellulosic waste types are distinguished by color. Black dashed lines represent the average value for each waste type. Softwood (SW); hardwood (HW); oriented strand board (OSB); medium density fiberboard (MDF).

Table 7
The cross-linkages among cellulose, hemicellulose, and lignin.

Cross-linkages	Types of bonds	Components
Intrapolymer	Ether	Lignin, hemicellulose, cellulose
	Ester	Hemicellulose
	Hydrogen	Cellulose
	Carbon to carbon	Lignin
Interpolymer	Hydrogen	Cellulose-hemicellulose
	Ether	Lignin-cellulose
	Ester	Lignin-hemicellulose

hemicellulose fragmentation [118]. The availability of lignin components was improved by alkali pretreatment, and thus the methane yield tended to increase with lignin content (Fig. 6C), which is consistent with the results of previous research [119]. Methane yield showed a negative correlation with lignin content, whatever the variation of I/S ratio (Fig. 6F). Yin et al. reported lignin inhibits the utilization of substrate (acetate) by bacteria in anaerobic digestion sludge, leading to lower methane production, and this inhibition effect enhanced with the increase of lignin content. Moreover, this inhibition could not be compensated by adding more acetate in the early stage [120]. In combination with microbiome studies in sludge, this situation may be due to the fact that the main microbial components in sludge do not have the ability to biodegrade lignin [121,122], therefore the availability of substrate decreases as lignin content increases.

4. Available biotechnology pretreatments for lignocellulosic wastes

The process of recovering energy from lignocellulosic waste generally involves pretreatment. There are many pretreatment technologies for lignocellulosic waste, which can be divided into physical pretreatment, chemical pretreatment, and biological pretreatment [123]. The purpose of all pretreatment technologies is to disintegrate lignin, cellulose, and hemicellulose as completely as possible, producing smaller fragments that are easily accessible to enzymatic hydrolysis or other biorefining processes for higher yields of added-value products (Fig. 7). In AD plants that consume lignocellulosic materials, the cost of the pretreatment process typically exceeds 40% of budget [9]. Although pretreatment technologies have been studied for many years and continuously improved, each method still suffers from obvious pitfalls in practice. For example, physical pretreatments such as grinding, steam

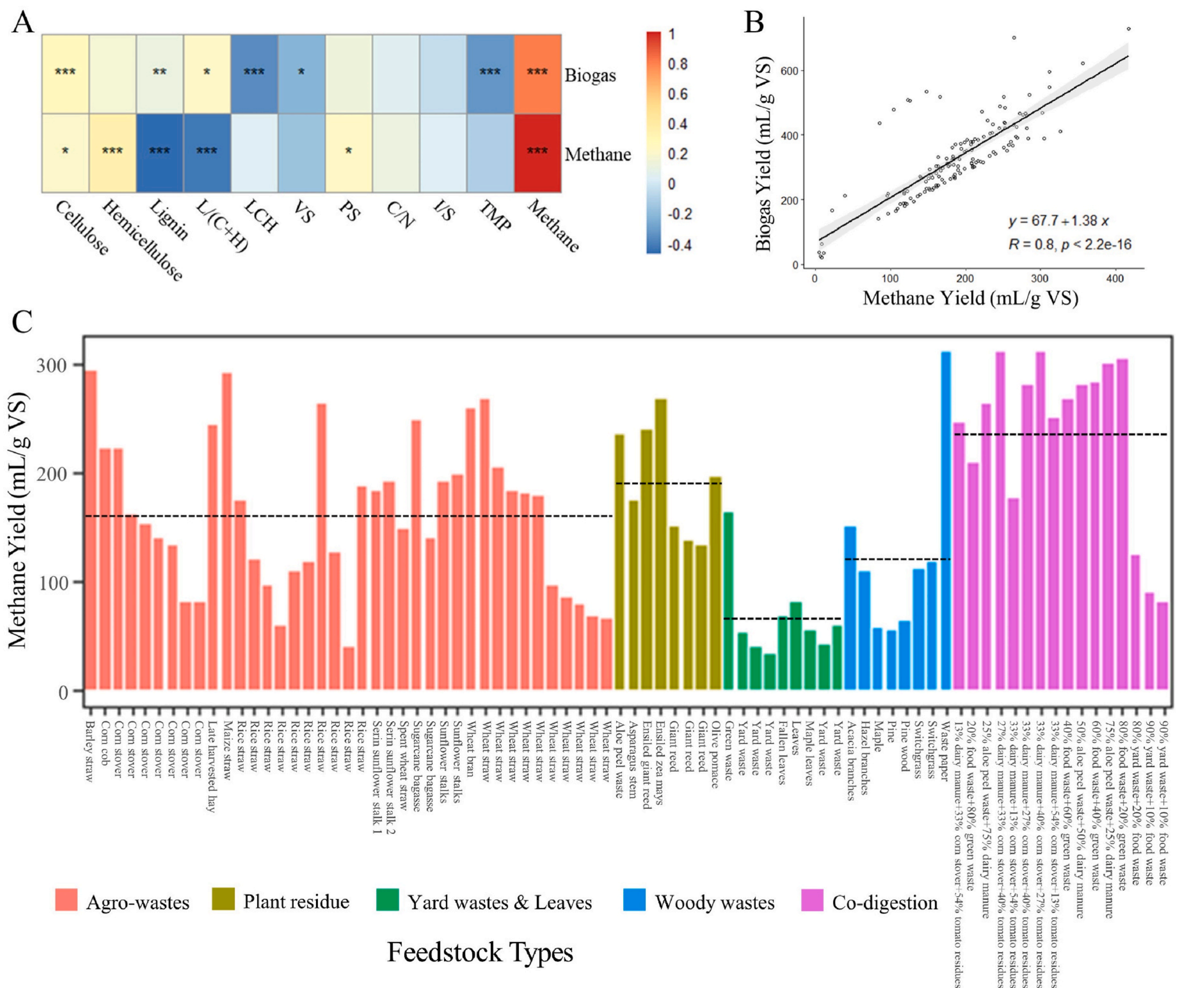


Fig. 5. (A) The correlation between parameters and product yield. (B) Curve fitting between methane and biogas yield. (C) Methane yield of different lignocellulose types (delete data containing pretreatment process). All figures are plotted from the data in Table B1. Lignocellulosic waste types are distinguished by color. Black dashed lines represent the average value for each waste type. L/(C + H): Lignin content/(Cellulose content + Hemicellulose content); LCH: Lignocellulose content (%) = Lignin content + Cellulose content + Hemicellulose content; VS: Volatile solids; PS: Particle size; I/S: inoculum (sludge) to substrate; TMP: theoretical methane potential (mL/g VS) were calculated according to the lignocellulose content [116].

explosion, ultrasound, microwave or thermal are energy-intensive and not cost efficient. Similarly, the application of chemicals like alkalis, acids or ionic liquids in pretreatment is faster but will generate wastewaters and toxic substances that require extra financial expenses for chemicals recycling. In contrast, the biological pretreatment, despite being a comparatively slower process, is a cost-effective technique that requires low energy input and is relatively free of hazardous chemicals [124,125]. However, the effect of biological pretreatment methods is currently not ideal due to limited technology. For example, the rapid and profitable production of cellulase has not yet been achieved. Nonetheless, scholars are increasingly interested in applying microorganisms or enzymes for pretreatment, with continuous attempts to screen suitable microbial communities with diverse enzymatic components and efficient hydrolysis activities.

In contrast to other techniques, biological pretreatments including enzymatic, fungal, and bacterial pretreatment, could be alternative options with low energy input and lower waste production. A broad range of microorganisms and their hydrolases found naturally can be used for biological pretreatment. In addition, there are many kinds of microorganisms that can degrade lignocellulose in the gut of xylophagous or saprophagous insects, so these insects offer a potential alternative for pretreatment. All in all, lignocellulose is difficult to be utilized as a low-nutrient resource, but many organisms have evolved systems in response to the fierce competition for food in nature that are capable of extracting value and energy. This section focuses on the status and development of biological pretreatment technologies for lignocellulosic waste recovery in recent years, especially those with potential applications. Examples of these pretreatments are described in Table 8, and the characteristics of these biotechnologies are shown in Fig. 7.

4.1. Lignocellulose degrading enzymes

To fully decompose lignocellulosic materials in pretreatment, the most direct and effective method is the application of enzymes because they can be selected specifically for the waste components, and the functions of different enzymes are affected by environmental conditions rather than affecting each other. There are three main types of enzymes, ligninolytic enzymes, cellulolytic enzymes and hemicellulolytic enzymes, that can be used depending on the composition of lignocellulosic waste [125]. Lignin is a complex and stubborn substance, and its degradation usually begins with an oxidation process, so ligninolytic enzymes include several oxidases including laccases, lignin peroxidase, manganese peroxidase and versatile peroxidase [150]. Fungi such as white rot fungi, brown rot and soft rot fungi are well-known producers of ligninolytic enzymes [151], while bacteria are considered to have low potential for lignin degradation. However, Bugg et al. believed bacteria

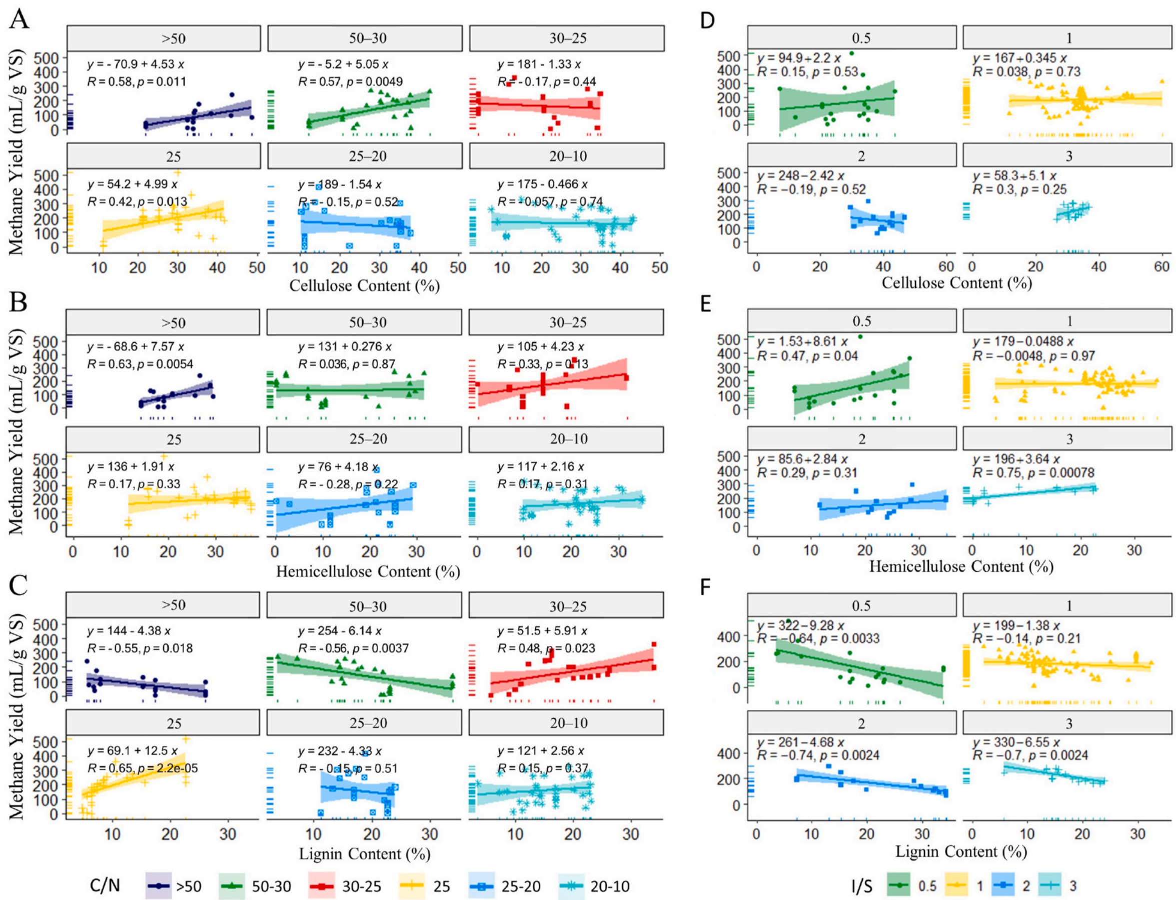


Fig. 6. Curve fitting between methane yield and feedstock lignocellulose composition under different C/N and I/S. A and D: Cellulose content (%); B and E: Hemicellulose content (%); C and F: Lignin content (%). All figures are plotted from the data in Table B1.

also play a role in producing ligninolytic enzymes and reviewed that actinomycetes, α-proteobacteria, and γ-proteobacteria are able to break down lignin [152]. Cellulolytic enzymes consist of endocellulases, exocellulases and cellobiases (β-glucosidases); these hydrolases can catalyze cellulose into monomeric sugar units through synergistic action. Endo- and exo-cellulases hydrolyze internal bonds in cellulose chains to release cellobiose, and β-glucosidases further cleaves cellobiose to glucose [153, 154]. Hemicellulolytic enzymes include glycoside hydrolases and carbohydrate esterases, where glycoside hydrolases hydrolyze glycosidic bonds and carbohydrate esterases hydrolyze ester linkages of ferulic acid or acetate groups [125]. These hemicellulolytic enzymes act synergistically in the hydrolysis of hemicellulose into several monomeric sugars [155]. Besides, polysaccharide monooxygenases discovered in recent years can assist cellulolytic and hemicellulolytic enzymes by oxidative disintegration of recalcitrant polysaccharide chains [156].

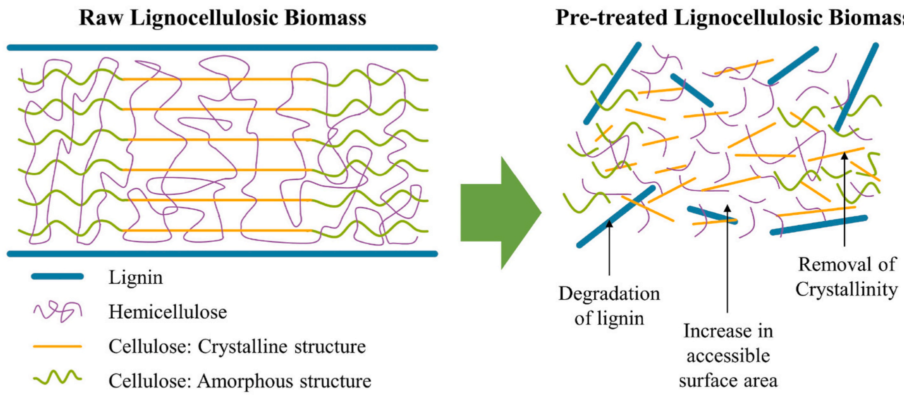
A number of studies focus on the application of enzymes for pretreatment. Kim et al. reported a technology for the pretreatment of livestock manure using pancreatin, secreting from the pancreas and containing a variety of digestive enzymes, and the results showed that hydrolysis efficiency and biogas yield were significantly improved [116]. Researchers invented two-stage combination strategies that combines enzymes with other techniques to improve the pretreatment process. Singh demonstrated a pretreatment technique combining chemical, physical or thermal approaches with enzymes widely used in lignocellulosic materials [157]. In another study, Wieczorek et al. found that enzymes (Onozuka R-10 and Macerozyme R-10) can facilitate the fermentation process of microalgal biomass resulting in an approximately 2-fold and 7-fold increase in methane and hydrogen yield, respectively [133].

Overall, other pretreatment techniques are required for lignocellulosic waste prior to enzymatic processing to increase hydrolysis rates and sugar yields. A successful high sugar yield enzymatic process requires a sterilization step to eliminate the consumption of released sugars by endogenous microorganisms prior to AD [158]. These additional steps could limit the industrial application of enzymatic pretreatment by imposing costs. Furthermore, the profitability of enzymatic pretreatment technologies essentially rely on the supply of low-cost enzyme sources. Although a variety of fungi (white rot fungi and brown rot fungi) have been investigated as an economical source of enzymes, the costs of commercial enzyme preparations are still high [159,160].

4.2. *Microbial community construction*

4.2.1. *Fungal communities*

Fungi are well known for their enzymatic degradation of lignocellulosic biomass, these fungi can produce various ligninolytic [161,162], cellulolytic [163], and hemicellulolytic enzymes [163]. Lignocellulolytic fungi include ascomycetes (*Aspergillus* sp., *Trichoderma reesei*), basidiomycetes (*Schizophyllum* sp., *Phanerochaete chrysosporium* and *Fomitopsis palustris*) and a few anaerobic species (*Orpinomyces* sp.) [125], of which studies have primarily focused on white-rot and brown-rot fungi [12]. White-rot fungi have been found to be the most effective in disintegration of lignin, because of various ligninolytic extracellular oxidases such as laccases, lignin peroxidases, versatile peroxidases, copper-dependent laccases and manganese peroxidases [164]. Some white-rot fungi, like *Ceriporiopsis subvermispora*, *Phellinus pini*, *Phlebia* sp. and *Pleurotus* sp., preferentially attack lignin rather than holocellulose



Biotechnology Pretreatments

Lignocellulose degrading enzymes	**Advantages:**	a) Lower incubation time b) Efficient hydrolysis of macromolecules
	Disadvantages:	a) Expensive in production and purification b) Poor stability of ligninolytic enzymes
Fungal communities	**Advantages:**	a) Low costs and less energy requirement b) Efficient degradation of lignin
	Disadvantages:	a) Long pretreatment time
Bacterial communities	**Advantages:**	a) Rapid growth and shorter pretreatment time
	Disadvantages:	a) Lower lignin degrading efficiency
Microbial co-culture systems	**Advantages:**	a) Various hydrolases for lignocellulose b) Enhanced productivity and biomass utilization
	Disadvantages:	a) Difficult to construct a stable functional system
Metabolically engineered microorganisms	**Advantages:**	a) Enhanced enzymatic activity b) Significant productivity improvement
	Disadvantages:	a) Limitation of industrial-scale application
Anaerobic co-digestion strategy	**Advantages:**	a) Nutrient balance and less inhibitors formation b) Enhanced productivity
	Disadvantages:	a) Loss of organic matter
Xylophagous or saprophagous animals	**Advantages:**	a) High economic effect b) Low costs for downstream recovery
	Disadvantages:	a) Less relevant research and immature technology

Fig. 7. The goals of pretreatment to overcome lignocellulose recalcitrant and characteristics of different biotechnologies.

(cellulose and hemicellulose) to delignify wood, leaving cellulose-rich materials. After delignification by white rot fungi, the decayed wood is whitish in color and fibrous in texture [165]. Unlike white-rot fungi, brown-rot fungi mainly utilize polysaccharides in lignocellulose while partially delignifying, resulting in shrinkage of the wood and a brown discoloration of oxidized lignin [166]. Other fungi degrade cell walls mainly through holocellulose, it is reported that *Trichoderma reesei* secretes xylanases and β-glucosidase which have high cellulase activities [125].

In a study of 30 bacterial and 18 fungal lignocellulolytic enzymes comparing their saccharification activities for lignocellulosic wastes [136], the overall activities of fungal hydrolases were more than twice that of bacteria. Fungal pretreatments are performed in an aerobic solid state fermentation system, requiring low bioreactor volume and less water. Results from several lab-scale studies investigating the effect of fungal pretreatment on BMP of various lignocellulosic wastes generally showed significant increases in methane yield, reaching 50% or higher, even in feedstocks with very low biodegradability [12]. Sindhu et al. found that fungal pretreatment could boost hydrolysis efficiency by 7–20 times, achieve 44–50% lignin removal, and completely eliminate any requirement for hazardous chemicals [9]. Although the lignin degradation pathway in fungi has been extensively studied since the late 1980s, there is as yet no breakthrough in its commercial application, mainly due to the practical challenges of bioengineering operability in complex fungal genomes.

4.2.2. Bacterial communities

Many bacteria secrete degrading enzymes that can decompose lignin, the selection of most efficient strains for pretreatment of lignocellulosic waste is the crucial step during biogas production. It is reported that DypB from *Rhodococcus jostii* RHA1 is a lignin peroxidase that is able to oxidize both polymeric lignin and a lignin model compound [132]. With advances in biotechnology, more and more bacteria have been identifying with the ability to degrade lignin through assays involving ^{14}C-labelled lignin, fluorescently labelled lignin or chemically nitrated lignin [167,168], and exist in three classes: actinomycetes, α-proteobacteria, and γ-proteobacteria [169]. Bugg et al. reviewed all four kinds of bacterial enzymes for lignin depolymerization, including Dyp-type peroxidase enzymes, multi-copper oxidase enzymes (laccases), manganese superoxide dismutase enzyme, and glutathione-dependent β-etherase enzymes [170]. By analyzing the genome sequences of 13 lignin-oxidizing bacteria, Granja-Travez et al. found that Dyp-type peroxidase enzymes and multi-copper oxidase enzymes are ubiquitous in the genomes of lignin-degrading bacteria, and the β-ketoadipate pathway appears to be the most common route for aromatic metabolism [171]. Unlike lignin, cellulose and hemicellulose are relatively easier to

Table 8
The biological pretreatments for valorization of lignocellulosic biomass.

Categories	Feedstocks	Biological technologies	Major benefits	Ref.
Fungi	Bamboo culms	*Punctularia* sp. TUFC20056	>50% lignin degradation	[126]
	Sawdust	*Pleurotus ostreatus*; *Pleurotus pulmonarius*	20 folds enhanced hydrolysis	[9]
	Wheat straw	*Ceriporiopsis subvermispora*	Enhanced hydrolysis	[127]
	Hardwood and straws	*Ceriporiopsis subvermispora*	Enhanced delignification	[128]
	Sugarcane bagasse	*Aspergillus niger*	High endoxylanase activity	[129]
Bacteria	Wood	*Petronet alfa* and *Petronet omega*	205% enhanced hydrolysis and 88.9% enhanced biomethane production	[130]
	Pine	*Agrobacterium* sp.	Enhanced delignification and 10 folds enhanced biomethane production	[131]
	Kraft lignin	*Rhodococcus jostii* RHA1	Enhanced delignification	[132]
Enzyme	Newspaper	Lignin peroxidase	Enhanced breakdown of lignocellulose and 41% enhanced biomethane production	[98]
	Livestock manure	Pancreas	Enhanced hydrolysis and biogas production	[116]
	Microalgae	Onozuka R-10; Macerozyme R-10	Enhanced hydrolysis efficiency	[133]
Co-culture	Sugarcane bagasse	*Clostridium thermocellum* + *Thermoanaerobacterium aotearoense*	Enhanced hydrolysis efficiency	[134]
	Corn stover	*Bacillus amyloliquefaciens* + *Trichoderma asperellum*	Enhanced hydrolysis efficiency	[135]
	Pine	Bacterial consortium	5–10 folds enhanced biogas production	[131]
	Sawdust	Microbial consortium	56.7% of lignin removal and 72.6% enhanced biomethane production	[52]
	Crop straws	Co-culturing of bacteria and fungus	3–6.6 folds enhanced hydrolysis	[136]
Metabolic engineering	Corn stover	Overexpression of Trvib-1 in *Trichoderma reesei* Rut-C30	Enhanced sugar release	[137]
	N/a	stk-12 deletion in *Neurospora crassa*	Enhanced cellulase production	[138]
	N/a	Xylose-isomerase pathway expression of *Saccharomyces cerevisiae*	Enhanced xylose utilization	[139]
	Corn stover	Vel1 overexpression of *Trichoderma asperellum*	Enhanced hydrolysis efficiency	[135]
Anaerobic co-digestion (AcoD)	Corn Stover	AcoD of swine manure	Enhanced biomethane yield	[140]
	Lignocellulosic biomass	AcoD of poultry feces	Enhanced biogas yield	[141]
	Lignocellulosic wastes	AcoD of food waste	Reduced hydraulic retention time and enhanced biomethane yield	[142]
	Straws	AcoD of bagasse	Enhanced biogas yield	[143]
	Grass	AcoD of forbs	Enhanced biomethane yield	[144]
	Agricultural wastes	AcoD of slaughter residues	Enhanced biomethane yield	[145]
	Different lignocellulosic wastes	AcoD of residual sludge	Enhanced biomethane yield	[146]
	Oat straw	AcoD of Tannery Wastes	60% enhanced biogas yield	[147]
Natural lignocellulose degrading systems	Corn silage and cattail	rumen fungus	Enhanced ch4 production	[148]
	Straws	Rumen bacteria	Enhanced hydrolysis and biogas yield	[37]
	Straws	Rumen	1.2–1.4 folds enhanced biogas production	[149]
	Paper, yard and wood	termite hindgut microorganism	49% enhanced biomethane production	[36]

degrade. For example, *Clostridium* and *Bacteroides* are capable of cellulose degradation [11], whilst *Thermomonospora fusca*, *Cellulomonas fimi* [125], various gram-positive strains (*Firmicutes*) and gram-negative strains (*Pseudomonas*, *Rahnella* and *Buttiauxella*) have been found to produce cellulases and show activity in degrading cellulosic materials [172].

Although bacterial lignin-degrading enzymes are less studied and characterized compared to fungi, with the maturity of omics technologies it is becoming clear that bacteria have a variety of degrading enzymes with potential in the pretreatment of lignocellulosic waste. Rashid et al. screened out a facultative anaerobic bacterium, *Agrobacterium* sp., from landfill soil with strong lignin degradation ability [131], then Muaaz-us-Salam et al. applied it in lignocellulosic waste pretreatment resulting in a 2-fold increase in biogas yield [98]. Bacteria are the main contributors to methane production during the anaerobic fermentation process, so the simplest strategy of biological pretreatment is to add bacteria for enhanced hydrolysis prior to anaerobic fermentation. Mirmohamadsadeghi et al. reported the impact of various pretreatments on anaerobic microbial diversity which can markedly affected biogas yield [82]. After applying different microbial inocula the microbial community in the fermentation process under the same conditions will remain different [173]. Microbial consortium TC-5, which included *Ruminiclostridium* (63%), *Thermoanaerobacterium* (23%), and *Caproiciproducens* (12%) and showed a variety of lignocellulolytic enzyme activity, was able to increase methane production by 22–36% during anaerobic digestion of untreated wheat straw [174].

4.2.3. *Microbial co-culture systems*

Due to the diverse composition of feedstocks, a complex enzymatic system is required for hydrolysis of lignocellulosic materials and methanogenesis fermentation processes. A single strain of any bacterium or fungus cannot simultaneously produce all required enzymes to achieve the desired effect, and microbial co-culture systems was thus proposed [175]. Although it is challenging for microorganisms with different degradation targets to survive and perform their functions together under the same culture medium, several co-culture systems have been found. A co-culture system is an ideal approach for one-step fermentation from lignocellulosic waste and offers several important advantages in constructing efficient bioproduction processes. Compared to microbial monoculture, co-culture systems can reduce the metabolic burden of each constituent strain and help to improve overall performance [176]. They also provide diversified cellular environments, which is conducive to the expression of hydrolases synthesized by different pathways and reduces the interference between different working modules [177,178]. There are three general co-culture systems in pretreatment process: bacterial co-culture, fungal co-culture, and bacterial and fungal co-culture [125].

In an example of bacterial co-culture systems, common cellulolytic bacteria are *Clostridium cellulovorans*, *Clostridium cellulolyticum*, and *Clostridium thermocellum*, that secrete various cellulases enzymes working together in cellulose hydrolysis [11]. Kato et al. also observed higher cellulose degradation in a mixed culture of *Clostridium straminisolvens* and three other aerobic non-cellulolytic bacteria [179]. Chandra et al.

found that *Paenibacillus* sp., *Aneurinibacillus aneurinilyticus*, and *Bacillus* sp. degraded kraft lignin by 37%, 33% and 30% respectively, while the mixed culture of these three bacteria reduced colour by 69%, lignin by 40% and total substrate by 50% under the same conditions [180]. Moreover, researchers experimented with mixed cultures of rumen bacteria, looking for possible combinations of high enzymatic activity to improve the degradation of lignocellulosic waste [181,182]. Culture of two or more species of fungi in pretreatment of lignocellulosic waste has been practiced for decades, with fungal co-cultures resulting in better enzymatic composition. There is considerable evidence for improved cellulolytic activity in fungal co-cultures. The mixed culture of *Trichoderma reesei* and *Aspergillus phoenicis* showed high levels of total cellulase and β-glucosidase production at 27 °C and pH of 5.5 [183]. Taha et al. compared the cellulase and xylanase activities of 30 isolated bacterial and 18 fungal strains by feeding four different straw substrates, then selected four fungal and five bacterial isolates based on their high enzymatic activities to construct dual and triple microbial panels. The results showed that fungal co-culture or bacterial co-culture increased hydrolysis efficiency by 2.8–6.6 times [136]. Furthermore, Chi et al. found high lignin degradation in the co-culture of *Ceriporiopsis subvermispora* and *Pleurotus ostreatus* [184]. Due to contrasting culture conditions (aerobic for fungi, anaerobic for lignin-degrading bacteria), few studies have focused on bacterial and fungal co-culture. In this co-culture pretreatment, fungi reduce the recalcitrance of lignin and hydrolyze holocellulose into soluble sugars, which are further converted by bacteria into valuable products.

However, finding suitable microbial co-culture systems is a long and arduous task because they have different genetic material, population size and ecological niches. Microorganisms performing different functions living in the same system may compete with each other for resources and fail to maximize their metabolic functions. At the same time, microbial communities living in natural habitats have extremely intricate food web networks, and it is difficult for most microorganisms to be cultivated artificially, which also reduces the available sources of microorganisms. Fortunately, the genetic makeup of bacteria enables them to rapidly adapt to new environments or to improve adaptive traits within their existing ecological niches [185]. Through domestication in highly competitive and hostile environments, combined with 16S rRNA gene sequencing, a co-culture system for efficient lignocellulose pretreatment can be obtained. Ali et al. isolated a novel microbial consortium using sterilized wood as a sole energy source, and biological pretreatment showed effective biodegradation in cellulose, hemicellulose, and lignin (37.5%, 39.6%, and 56.7%, respectively). Pretreatment increased biogas production by 72.6% after 28 days of anaerobic fermentation [52].

Overall, microbial co-culture systems for pretreatment have the potential to be combined with anaerobic digestion. However, detailed biochemical characterization of the various co-culture microorganisms involved in the conversion of lignocellulose to biogas is required to construct compatible and efficient systems for both pretreatment and anaerobic digestion processes.

4.2.4. Metabolically engineered microorganisms

In general, microbial species used for pretreatment have been improved through random mutagenesis and screening processes, but these processes are largely slow, unpredictable and limited by microbial intrinsic phenotype. Therefore, metabolic engineering for purposeful enhancement of microbes attracts considerable attention [186]. To boost biogas production, a comprehensive understanding of the metabolic pathways of the microbial communities is the primary prerequisite. High throughput metagenomic sequencing is used to identify the genetic components of various microorganisms from different environments that contribute significantly to biofuel production, and bioinformatics tools and gene editing techniques enable targeted microbial improvements to boost biogas production [187,188].

Metabolic engineering can provide microorganisms with higher productivity and product diversity. Overexpression of *Trvib-1* in *Trichoderma reesei* increased its cellulase production by 200%, while a 40% increase in corn stover hydrolysis was observed [137]. Karuppiah et al. reported that by overexpressing *Vel1* in *Trichoderma asperellum* and co-culturing it with *Bacillus amyloliquefaciens*, higher enzymatic activity and 90% hydrolysis efficiency were achieved [135]. Moreover, the process of biofuels recovery from certain lignocellulosic waste can be simplified through metabolic engineering. Rollin et al. combined more than 10 purified enzymes into artificial enzymatic pathways, achieving high yields from glucose and xylose to hydrogen [189]. Furthermore, metabolic engineering allows microorganisms to utilize cheap feedstocks, reducing the cost of fuel production. *Saccharomyces cerevisiae* cannot grow on xylose and thus cannot efficiently utilize lignocellulosic waste [190], thus many studies have improved the xylose utilization rate of *Saccharomyces cerevisiae* through metabolic engineering [190].

In addition to enhancing the hydrolysis ability of the microorganism itself through genetic modification, it is also an effective approach to introduce the genes encoding lignocellulose hydrolase into non-hydrolyzed microorganisms of high growth yield [187]. Sedlak et al. introduced three xylose metabolizing genes (xylose reductase, xylitol dehydrogenase and xylulokinase) into *Saccharomyces* yeast through metabolic engineering to enable glucose and xylose co-fermentation [191]. Divia et al. used recombinant DNA technology to develop anaerobic bacteria with enhanced hydrolysis properties that could reduce hydraulic retention time and increase biogas yield [192].

Although metabolic engineering can certainly enhance beneficial characteristics such as hydrolysis performance of microorganisms to increase methane production, industrial-scale applications of genetically modified strains may be limited by the regulations of genetically modified organisms (GMO) [193]. In view of this, isolation of naturally hydrolyzing microorganisms and screening for suitable microbial co-culture systems are still considered a promising tool for enhancing biogas production.

4.3. Anaerobic co-digestion strategy

The high carbon content in lignocellulosic wastes hinders the optimal C/N ratio required for anaerobic digestion. Therefore, it is necessary to increase the nitrogen content of lignocellulosic wastes prior to anaerobic digestion, and protein-rich wastes including food wastes and animal manures are ideal candidates to compensate for nitrogen deficiency in lignocellulosic wastes. Besides increasing the availability of feedstocks, anaerobic co-digestion (a fermentation process of two or more substrates to produce biogas) also provides the following benefits: nutrient balance (good for microbial growth); dilution of inhibitory or toxic substances (high water content in food waste or animal manure); higher load of biodegradable organic matter; and increasing methane production per unit volume [194]. Several studies have shown that methane production from lignocellulosic wastes and process performance of anaerobic digestion can be enhanced by co-digestion with various substrates. Data from the Web of Science showed that 91.8% of all publications in lignocellulosic waste co-digestion was published after 2015. Of these, animal manures were the most widely used co-substrate (38.5%), followed by food wastes (33.5%), sewage sludge (20.3%) and other co-substrates such as slaughterhouse wastes, municipal wastes and agricultural residues (7.6%).

Anaerobic digestion can be broadly classified into dry (>15% dry matter content) or wet (<15% dry matter content) processes [195]. Due to the poor fluidity and the complex structure, lignocellulosic wastes are not easily decomposed by enzymes or anaerobic microorganisms, resulting in low efficiency in practice. However, this low efficiency can be improved by adding co-substrates to improve the composition balance. Co-digestion is commonly a wet single step processes, which allows simpler mechanical handling of wastes and allows the system to be carried out in a continuous flow stirred tank reactor (CSTR) [194]. Alvarez and Liden improved methane production by wet anaerobic co-digestion, in which livestock manure was added to quinoa stems [196]. The biodegradability of feedstocks can be enhanced by the synergistic effect of co-substrates. Wang et al. studied the effect of different mixture components of dairy manure, chicken manure and wheat straw on methane production and found that better performance of anaerobic co-digestion can be fulfilled by optimizing the feed composition and C/N ratio [197]. Similarly, the methane yield in co-digestion of cow manure and straw was significantly higher than that of mono-component anaerobic digestion [198]. Moreover, co-digestion can avoid the phenomenon of continuously increasing substrate levels during single-substrate digestion which inhibits system operation. Pages-Diaz et al. found that the anaerobic co-digestion of slaughterhouse wastes with agricultural residues did not result in the suppression of methanogenic communities compared to single digestion of slaughterhouse wastes. In the case of single digestion of slaughterhouse waste, the release of high levels of long-chain fatty acids can inhibit the fermentation process [199].

Although there are numerous reports about co-digestion of lignocellulosic material improving methane production, some studies obtained lower specific methane production during co-digestion compared to single digestion [200,201]. Since seemingly ideal co-digestion wastes also confer antagonistic effects, comprehensive and continuous experiments are recommended. It is important to characterize and evaluate the co-digestion effect before applying it to industrialization. In conclusion, anaerobic co-digestion has a positive synergistic effect on process stability and methane production, increasing the biodegradable components, facilitating contact of microorganisms or enzymes with substrates, and enriching the microbial community [194]. Successful co-digestion of lignocellulosic material with other wastes is a potentially economical process that converts lignocellulosic wastes into biogas while leaving a nutrient-rich residue that can be used as fertilizer.

4.4. Other xylophagous or saprophagous animals

Some xylophagous or saprophagous animals, like diverse taxonomic groups of insects (termites, beetles, black soldier fly etc.), earthworms and ruminant animals, also have a strong ability to degrade lignocellulosic wastes [202]. These organisms can synergistically decompose lignocellulosic biomass through a series of physiological mechanisms, including physiological mechanisms, enzymatic, and intestinal flora. Firstly, the gut or stomach of these organisms has bacteria, archaea, fungi, protozoa and different enzymatic components that efficiently digest lignocellulose. Secondly, they have specific feeding/chewing mechanisms that help physically break down lignocellulose into fine particles for subsequent adequate degradation. Finally, these organisms have developed a well-established fermentation system, and a series of inhibitory compounds produced during the fermentation process can be transferred so as not to affect the efficiency of the system.

Termites are capable of efficiently digesting lignocellulosic materials such as wood. It has been reported that there are highly expressed cellulase genes in the gut of termites [203]. In addition, genes encoding lignin-degrading enzymes and protection from reactive oxygen species and other toxic metabolites produced during lignin degradation were found in its genome [204,205]. Some xylophagous or saprophagous insects also have the potential to degrade lignocellulosic wastes. For example, Gao et al. found that black soldier flies can directly use crop straw as food [61], and some beetles have been reported to survive in wood [204]. Like termite gut microbiology, rumen microbiology has been extensively studied and applied to lignocellulosic material degradation based on its abundant internal lignocellulose degrading enzymes [206]. There is a growing interest in the use of rumen microorganisms for anaerobic digestion of lignocellulosic wastes such as crop straws [149,207], wood or sawdust [208], municipal solid waste [209]. Composting can also be used as a pretreatment before anaerobic digestion. The temperature of raw materials is increased by composting prior to biogas production, thereby reducing heat requirements for subsequent start-up of the digestion process [12].

It is worth noting that pretreatment with these macroorganisms requires an aerobic environment due to survival requirements. As Tabatabaei et al. stated microaeration pretreatment is technically more efficient and cost-effective [187]. It has long been believed that the presence of oxygen is necessary for lignin degradation, and anaerobic conditions are not conducive to lignin degradation, although the mechanism is unclear [202]. By controlling oxygen levels, providing a growth environment for many facultative bacteria, higher hydrolysis rates can be achieved to accelerate methane production [210]. On the other hand, aerobic pretreatment may lead to loss of organic matter (feeding consumption of organisms) and consequently decrease biogas yield, thus reducing oxygen content could partially avoid organic biomass loss. Last but not least, the residual oxygen in system after aerobic pretreatment does not interfere with biogas production process, as methanogens have been identified to have specific mechanisms in response to micro-aeration conditions and maintain their function with little or no inhibition [211,212].

5. Economic aspects of biogas production from lignocellulosic waste

In the economic chain of waste manipulation, many factors determine waste management costs, in which collection and transportation to processing facilities can be important factors. Lignocellulose wastes are irregular in size and difficult to store and transport, however compacting and pressing could reduce lignocellulosic waste volume and improve manipulation, leading to lower transportation costs [213]. It is difficult to put a price on the collection and transportation of MSW because of the heterogeneous and complex composition. As it is not commonly collected, forestry waste does not currently have a price in the open market. Therefore, only an estimated price could be provided, ranging from 18 to 50 UK £ per oven dry tons [214]. The characteristics of crop straw, low density and high moisture content, make their collection, transportation and storage in large quantity rather difficult. It is reported that the cost of rice straw delivered to an industrial site in Punjab is about 864.24 INR (about £9.5) per ton, including payment to farmer, baling, transportation, and storage [215]. In general, the costs mainly depend on these criteria: transport distance, storage and drying, type and size of machinery used, steepness of the terrain, and labor costs [216]. The collection and transportation of lignocellulosic waste to the biogas production plant could be economically achieved by introducing appropriate mechanization hardware and practices, in the form of larger and more efficient baling, handling, and transport equipment will result in lower costs. On the other hand, with industrialization and scale-up, biogas production plants could be set up at the source of lignocellulosic waste to significantly reduce transportation costs.

Anaerobic digestion producing biogas is one of the most promising options for lignocellulosic wastes management and valorization, as it contributes to the entire circular economy chain. It is a convenient and generally cost-effective technology that satisfies efficient disposal of waste and energy production, with possible resources recovery from digestion residues. Fig. 8 shows the pathways of biogas production from lignocellulosic wastes and its application. Part of the organic matter in industrial biogas plants remains in the solid phase of the digestate, which can be then separated and used as fertilizer [217]. Biogas generally refers to a gas mixture consisting mainly of methane (55–65%), carbon dioxide (30–35%) and other trace gases, like hydrogen sulphide [218]. Biogas from anaerobic digestion could be combusted directly for cooking or used for power generation, which emits less GHG than fossil fuels [219]. However, the presence of CO_2 in biogas limits its calorific value due to its incombustibility, thus limiting its applicability and transportability. In addition, trace amounts of H_2S could corrode equipment such as generators and diesel engines. Therefore, biogas needs to be upgraded to be used as a vehicle fuel [200], and the upgrading process generates a highly concentrated CO_2 stream

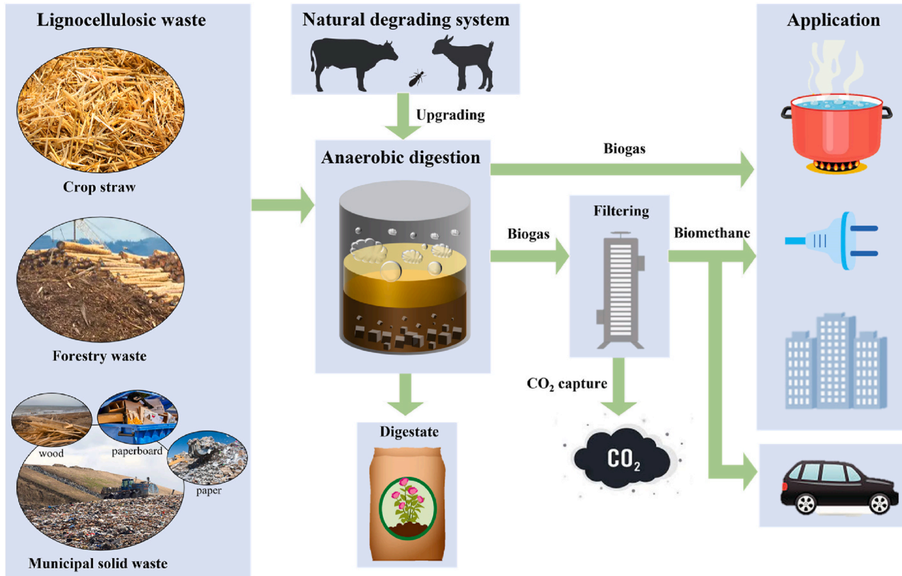


Fig. 8. Schematic diagram of biogas production pathways from lignocellulosic wastes.

leading to CO_2 capture costs as low as 20 dollars per ton [220]. Carbon prices strengthen the economic case for biogas consumption, facilitating anaerobic digestion of lignocellulosic wastes, and providing rural communities with an additional source of income.

A total of 430 biogas plants worldwide were registered with the International Energy Agency (IEA) by the end of 2015. According to EPA statistics, biogas usage will reach 14 EJ in 2050, which plays an important role in how the global energy sector can reach net-zero emissions by 2050. In addition, household and village digesters in rural areas will provide nearly 500 million households with renewable energy and clean cooking by 2030 [2]. Biogas is a consolidated market with a positive outlook, and with global policies leaning toward sustainable new energy, investment in low-carbon gases such as biogas and biomethane will rise to 14 billion dollars by 2040 [221]. For import-dependent countries, investment in biomethane supplies can replace the need for fuel imports. For example, China and India both have extensive biomethane potential, a large portion of which can be obtained at relatively low cost [59,60]. If biomethane replaces natural gas needs, the two countries would save tens of billions in import bills each year, which could help offset the cost of developing a domestic biomethane industry.

Currently, about 30 Mtoe of biomethane can be developed at a lower cost than natural gas. Methane contributes significantly to the greenhouse effect, and if policies recognize the value of avoiding methane emissions from the decomposition of feedstocks, larger biomethane production will be cost-competitive. Overall, biogas and biomethane have great economic potential in promoting the clean energy transition and achieving the sustainable development goals.

6. Perspectives and conclusions

In this review article, many biological techniques, which appear to be a promising process for the valorization of lignocellulosic wastes, are summarized. Although this review indicates that biological pretreatments are energy saving and economical, more research is still needed to address several major challenges: (1) technical challenges related to biological techniques. Anaerobic co-digestion, microbial coculture and metabolically engineered microorganisms are promising, but there are still limitations to maximize the value of lignocellulosic waste. In the future, it is possible to consider improving single step to multi-step pretreatment while integrating other advantageous technologies. (2) modification of anaerobic digestion. Based on the complex structure of lignocellulosic wastes and the instability of anaerobic digestion, it is necessary to optimize operating conditions such as temperature, pH, fermentation time, solids loading, and hydraulic retention time. On the other hand, improving anaerobic reactors by imitating AD in natural ecosystems is of interest. For example, the digestion strategies of other animal systems (e.g. rumen and termite gut) could be beneficial to the design and construction of anaerobic reactors, from simple batch reactors to complex continuous reactor systems to facilitate AD of lignocellulosic biomass. (3) kinetics and modeling studies. Understanding the fundamentals for all factors that affect the biological processes is important for improving the potential for biogas production from lignocellulosic wastes. These models can be enhanced by more experimental data and collaboration of research scholars, including evaluating the potential of different lignocellulosic wastes, developing kinetic models for lignocellulose degradation, helping to optimize the pretreatment process, describing anaerobic digestion process and finding optimal fermentation conditions.

National and regional governments have an opportunity to develop circular economy principles through the introduction of policies to support the production of biogas from lignocellulosic wastes, for example through maximizing the availability of feedstocks and supporting biogas and biomethane supply and consumption. Firstly, governments may introduce comprehensive waste management policies and regulations to strengthen the collection and classification of lignocellulosic wastes creating feedstocks for biogas production. Secondly, government should promote investment, especially in remote areas where lignocellulosic wastes are abundant, adopting strategies of distributed biogas production to avoid long-distance transportation costs to centralized large-scale biogas power plants. Finally, options include introducing usage incentives (lower usage prices) for supporting consumption, and establishing biomethane standards for injecting to the gas network and direct use as transport energy.

AD is a reliable technology for biogas production from lignocellulosic wastes, and biological techniques, which are reviewed in this article, could be vital for the valorization of lignocellulosic wastes. Using biological techniques to improve the overall efficiency of anaerobic digestion has tremendous applications for sustainability in the future. Although many biotechnologies have been highly developed, detailed experimentation is still required to optimize procedure based on kinetic models. In addition, the amount of lignocellulosic wastes generated each year is enormous, and only a small proportion is treated in a manner to reduce greenhouse gas emissions. In this regard, biogas and biomethane generated from lignocellulosic waste have great economic and environmental potential. Combined with government policy support, a system that integrates lignocellulosic wastes recycling, minimizes emissions, produces clean bioenergy, and is economically viable can be successful worldwide.

Declaration of competing interest

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

Data availability

No data was used for the research described in the article.

Acknowledgements

We gratefully acknowledge financial support from the China Scholarship Council (CSC) for Zhenghui Gao (CSC No. 202006760088), Yuan Li (CSC No. 202104910427), and Hang Qian (CSC No. 202106760089), and the financial support from the Saudi Arabian Ministry of Education for Khaled Alshehri.

Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.rser.2022.112995.

Appendix A. Supplementary data

The following is the supplementary data related to this review.

References

[1] Nations U. The role of fossil fuels in a sustainable energy system. U N n.d. https://www.un.org/en/chronicle/article/role-fossil-fuels-sustainable-energy-system (accessed February 21, 2022).
[2] Outlook for biogas and biomethane: prospects for organic growth – Analysis. IEA n.d. https://www.iea.org/reports/outlook-for-biogas-and-biomethane-prospects-for-organic-growth (accessed February 24, 2022).
[3] Roy R, Rahman MS, Raynie DE. Recent advances of greener pretreatment technologies of lignocellulose. Curr Res Green Sustain Chem 2020;3:100035. https://doi.org/10.1016/j.crgsc.2020.100035.
[4] Covey KR, Megonigal JP. Methane production and emissions in trees and forests. New Phytol 2019;222:35–51. https://doi.org/10.1111/nph.15624.
[5] Obulisamy PK, , Sim Yan May J, Rajasekar B. Gradient packing bed bio-filter for landfill methane mitigation. Bioresour Technol 2016;217:205–9. https://doi.org/10.1016/j.biortech.2016.01.059.
[6] Olatunji KO, Ahmed NA, Ogunkunle O. Optimization of biogas yield from lignocellulosic materials with different pretreatment methods: a review. Biotechnol Biofuels 2021;14:159. https://doi.org/10.1186/s13068-021-02012-x.
[7] Millati R, Wikandari R, Ariyanto T, Putri RU, Taherzadeh MJ. Pretreatment technologies for anaerobic digestion of lignocelluloses and toxic feedstocks. Bioresour Technol 2020;304:122998. https://doi.org/10.1016/j.biortech.2020.122998.
[8] Sun J, Zhang L, Loh K-C. Review and perspectives of enhanced volatile fatty acids production from acidogenic fermentation of lignocellulosic biomass wastes. Bioresour Bioprocess 2021;8:68. https://doi.org/10.1186/s40643-021-00420-3.
[9] Sindhu R, Binod P, Pandey A. Biological pretreatment of lignocellulosic biomass – an overview. Bioresour Technol 2016;199:76–82. https://doi.org/10.1016/j.biortech.2015.08.030.
[10] Zabed HM, Akter S, Yun J, Zhang G, Awad FN, Qi X, et al. Recent advances in biological pretreatment of microalgae and lignocellulosic biomass for biofuel production. Renew Sustain Energy Rev 2019;105:105–28. https://doi.org/10.1016/j.rser.2019.01.048.
[11] Cheng H-H, Whang L-M. Resource recovery from lignocellulosic wastes via biological technologies: advancements and prospects. Bioresour Technol 2022;343:126097. https://doi.org/10.1016/j.biortech.2021.126097.
[12] Carrere H, Antonopoulou G, Affes R, Passos F, Battimelli A, Lyberatos G, et al. Review of feedstock pretreatment strategies for improved anaerobic digestion: from lab-scale research to full-scale application. Bioresour Technol 2016;199:386–97. https://doi.org/10.1016/j.biortech.2015.09.007.
[13] Zabed H, Sultana S, Sahu JN, Qi X. An overview on the application of ligninolytic microorganisms and enzymes for pretreatment of lignocellulosic biomass. In: Sarangi PK, Nanda S, Mohanty P, editors. Recent adv. Biofuels bioenergy util. Singapore: Springer; 2018. p. 53–72. https://doi.org/10.1007/978-981-13-1307-3_3.
[14] Zhang Y-HP. Reviving the carbohydrate economy via multi-product lignocellulose biorefineries. J Ind Microbiol Biotechnol 2008;35:367–75. https://doi.org/10.1007/s10295-007-0293-6.
[15] Samuel R, Pu Y, Raman B, Ragauskas AJ. Structural characterization and comparison of switchgrass ball-milled lignin before and after dilute acid pretreatment. Appl Biochem Biotechnol 2010;162:62–74. https://doi.org/10.1007/s12010-009-8749-y.
[16] Rao MN, Sultana R, Kota SH. Chapter 2 - municipal solid waste. In: Rao MN, Sultana R, Kota SH, editors. Solid hazard. Waste manag. Butterworth-Heinemann; 2017. p. 3–120. https://doi.org/10.1016/B978-0-12-809734-2.00002-X.
[17] Hoornweg D, Bhada-Tata P. What a waste : a global review of solid waste management. Washington, DC: World Bank; 2012.
[18] Us Epa O. National overview: facts and figures on materials, wastes and recycling. 2017 (accessed March 2, 2022), https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/national-overview-facts-and-figures-materials.
[19] National Bureau of Statistics of China. China statistical yearbook. 2019 (accessed March 2, 2022), http://www.stats.gov.cn/tjsj/ndsj/2019/indexch.htm.
[20] Oecd Statistics n.d (accessed March 2, 2022), https://stats.oecd.org/.
[21] Us Epa O. Facts and figures about materials, waste and recycling. 2017 (accessed March 2, 2022), https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling.
[22] Overview - waste - Eurostat n. d. https://ec.europa.eu/eurostat/web/waste/overview.
[23] Ferronato N, Torretta V. Waste mismanagement in developing countries: a review of global issues. Int J Environ Res Publ Health 2019;16:1060. https://doi.org/10.3390/ijerph16061060.
[24] Besserer A, Troilo S, Girods P, Rogaume Y, Brosse N. Cascading recycling of wood waste: a review. Polymers 2021;13:1752. https://doi.org/10.3390/polym13111752.
[25] Dexter M, Rickman K, Pan C, Chang C, Malhotra R. Intense Pulsed Light unprinting for reducing life-cycle stages in recycling of coated printing paper. J Clean Prod 2019;232:274–84. https://doi.org/10.1016/j.jclepro.2019.05.387.
[26] Statista - the statistics portal. Statista n.d. https://www.statista.com/markets/408/topic/435/waste-management//(accessed March 7, 2022).
[27] Ding Y, Zhao J, Liu J-W, Zhou J, Cheng L, Zhao J, et al. A review of China's municipal solid waste (MSW) and comparison with international regions: management and technologies in treatment and resource utilization. J Clean Prod 2021;293:126144. https://doi.org/10.1016/j.jclepro.2021.126144.
[28] Khan S, Anjum R, Raza ST, Ahmed Bazai N, Ihtisham M. Technologies for municipal solid waste management: current status, challenges, and future perspectives. Chemosphere 2022;288:132403. https://doi.org/10.1016/j.chemosphere.2021.132403.
[29] Millati R, Cahyono RB, Ariyanto T, Azzahrani IN, Putri RU, Taherzadeh MJ. Chapter 1 - agricultural, industrial, municipal, and forest wastes: an overview. In: Taherzadeh MJ, Bolton K, Wong J, Pandey A, editors. Sustain. Resour. Recovery zero waste approaches. Elsevier; 2019. p. 1–22. https://doi.org/10.1016/B978-0-444-64200-4.00001-3.
[30] Alfaia RG de SM, Costa AM, Campos JC. Municipal solid waste in Brazil: a review. Waste Manag Res 2017;35:1195–209. https://doi.org/10.1177/0734242X17735375.
[31] Cragg SM, Beckham GT, Bruce NC, Bugg TD, Distel DL, Dupree P, et al. Lignocellulose degradation mechanisms across the tree of life. Curr Opin Chem Biol 2015;29:108–19. https://doi.org/10.1016/j.cbpa.2015.10.018.
[32] O'Dwyer J, Walshe D, Byrne KA. Wood waste decomposition in landfills: an assessment of current knowledge and implications for emissions reporting. Waste Manag 2018;73:181–8. https://doi.org/10.1016/j.wasman.2017.12.002.
[33] Ufarté L, Potocki-Veronese G, Cecchini D, Tauzin AS, Rizzo A, Morgavi DP, et al. Highly promiscuous oxidases discovered in the bovine rumen microbiome. Front Microbiol 2018;9:861. https://doi.org/10.3389/fmicb.2018.00861.
[34] Schroyen M, Van Hulle SWH, Holemans S, Vervaeren H, Raes K. Laccase enzyme detoxifies hydrolysates and improves biogas production from hemp straw and miscanthus. Bioresour Technol 2017;244:597–604. https://doi.org/10.1016/j.biortech.2017.07.137.
[35] Schroyen M, Vervaeren H, Vandepitte H, Van Hulle SWH, Raes K. Effect of enzymatic pretreatment of various lignocellulosic substrates on production of phenolic compounds and biomethane potential. Bioresour Technol 2015;192:696–702. https://doi.org/10.1016/j.biortech.2015.06.051.
[36] Rahimi H, Sattler ML, Hossain MDS, Rodrigues JLM. Boosting landfill gas production from lignin-containing wastes via termite hindgut microorganism. Waste Manag 2020;105:299–308. https://doi.org/10.1016/j.wasman.2020.02.007.
[37] Dollhofer V, Dandikas V, Dorn-In S, Bauer C, Lebuhn M, Bauer J. Accelerated biogas production from lignocellulosic biomass after pre-treatment with Neocallimastix frontalis. Bioresour Technol 2018;264:219–27. https://doi.org/10.1016/j.biortech.2018.05.068.
[38] Sanitha M, Aliya Fathima A, Ramya M. Microbial diversity analysis of wood degrading microbiome and screening of natural consortia for bioalcohol production. Biofuels 2021;12:697–702. https://doi.org/10.1080/17597269.2018.1532751.
[39] Ranganathan A, Smith OP, Youssef NH, Struchtemeyer CG, Atiyeh HK, Elshahed MS. Utilizing anaerobic fungi for two-stage sugar extraction and biofuel production from lignocellulosic biomass. Front Microbiol 2017;8:635. https://doi.org/10.3389/fmicb.2017.00635.

[40] Hardersen S, Zapponi L. Wood degradation and the role of saproxylic insects for lignoforms. Appl Soil Ecol 2018;123:334–8. https://doi.org/10.1016/j.apsoil.2017.09.003.
[41] Grassi G, House J, Dentener F, Federici S, den Elzen M, Penman J. The key role of forests in meeting climate targets requires science for credible mitigation. Nat Clim Change 2017;7:220–6. https://doi.org/10.1038/nclimate3227.
[42] Pan Y, Birdsey RA, Fang J, Houghton R, Kauppi PE, Kurz WA, et al. A large and persistent carbon sink in the world's forests. Science 2011;333:988–93. https://doi.org/10.1126/science.1201609.
[43] Faostat nd (accessed March 6, 2022), https://www.fao.org/faostat/en/#data/FO.
[44] Russell MB, Fraver S, Aakala T, Gove JH, Woodall CW, D'Amato AW, et al. Quantifying carbon stores and decomposition in dead wood: a review. For Ecol Manag 2015;350:107–28. https://doi.org/10.1016/j.foreco.2015.04.033.
[45] Tatti D, Fatton V, Sartori L, Gobat J-M, Le Bayon R-C. What does 'lignoform' really mean? Appl Soil Ecol 2018;123:632–45. https://doi.org/10.1016/j.apsoil.2017.06.037.
[46] Saunois M, Stavert AR, Poulter B, Bousquet P, Canadell JG, Jackson RB, et al. The global methane budget 2000–2017. Earth Syst Sci Data 2020;12:1561–623. https://doi.org/10.5194/essd-12-1561-2020.
[47] Le Quéré C, Andres RJ, Boden T, Conway T, Houghton RA, House JI, et al. The global carbon budget 1959–2011. Earth Syst Sci Data 2013;5:165–85. https://doi.org/10.5194/essd-5-165-2013.
[48] Global FAO. Forest resources assessment 2020: main report. Rome, Italy: FAO; 2020. https://doi.org/10.4060/ca9825en.
[49] Lee E, Han H-S. Air curtain burners: a tool for disposal of forest residues. Forests 2017;8:296. https://doi.org/10.3390/f8080296.
[50] Nicholls DL, Halbrook JM, Benedum ME, Han H-S, Lowell EC, Becker DR, et al. Socioeconomic constraints to biomass removal from forest lands for fire risk reduction in the western U.S. Forests 2018;9:264. https://doi.org/10.3390/f9050264.
[51] Amirta R, Herawati E, Suwinarti W, Watanabe T. Two-steps utilization of shorea wood waste biomass for the production of oyster mushroom and biogas – a zero waste approach. Agric Agric Sci Procedia 2016;9:202–8. https://doi.org/10.1016/j.aaspro.2016.02.127.
[52] Ali SS, Abomohra AE-F, Sun J. Effective bio-pretreatment of sawdust waste with a novel microbial consortium for enhanced biomethanation. Bioresour Technol 2017;238:425–32. https://doi.org/10.1016/j.biortech.2017.03.187.
[53] Akyol Ç, Ince O, Bozan M, Ozbayram EG, Ince B. Fungal bioaugmentation of anaerobic digesters fed with lignocellulosic biomass: what to expect from anaerobic fungus Orpinomyces sp. Bioresour Technol 2019;277:1–10. https://doi.org/10.1016/j.biortech.2019.01.024.
[54] Oh J-I, Lee J, Lin K-YA, Kwon EE, Fai Tsang Y. Biogas production from food waste via anaerobic digestion with wood chips. Energy Environ 2018;29:1365–72. https://doi.org/10.1177/0958305X18777234.
[55] Kim S, Dale BE. Global potential bioethanol production from wasted crops and crop residues. Biomass Bioenergy 2004;26:361–75. https://doi.org/10.1016/j.biombioe.2003.08.002.
[56] Yan B, Yan J, Li Y, Qin Y, Yang L. Spatial distribution of biogas potential, utilization ratio and development potential of biogas from agricultural waste in China. J Clean Prod 2021;292:126077. https://doi.org/10.1016/j.jclepro.2021.126077.
[57] Faostat nd (accessed March 15, 2022), https://www.fao.org/faostat/en/#data/QCL.
[58] Nguyen TT, Nguyen VC. Financial development and renewables in southeast asian countries—the role of organic waste materials. Sustainability 2021;13:8748. https://doi.org/10.3390/su13168748.
[59] Zhao L, Meng H, Shen Y, Ding J, Zhang X. Investigation and analysis of planting-breeding circulating agriculture ecosystem system in northern plains in China. Trans Chin Soc Agric Eng 2017;33:1–10.
[60] Kapoor R, Ghosh P, Kumar M, Sengupta S, Gupta A, Kumar SS, et al. Valorization of agricultural waste for biogas based circular economy in India: a research outlook. Bioresour Technol 2020;304:123036. https://doi.org/10.1016/j.biortech.2020.123036.
[61] Gao Z, Wang W, Lu X, Zhu F, Liu W, Wang X, et al. Bioconversion performance and life table of black soldier fly (Hermetia illucens) on fermented maize straw. J Clean Prod 2019;230:974–80. https://doi.org/10.1016/j.jclepro.2019.05.074.
[62] Bhuvaneshwari S, Hettiarachchi H, Meegoda JN. Crop residue burning in India: policy challenges and potential solutions. Int J Environ Res Publ Health 2019;16:832. https://doi.org/10.3390/ijerph16050832.
[63] Sawlani R, Agnihotri R, Sharma C, Patra PK, Dimri AP, Ram K, et al. The severe Delhi SMOG of 2016: a case of delayed crop residue burning, coincident firecracker emissions, and atypical meteorology. Atmos Pollut Res 2019;10:868–79. https://doi.org/10.1016/j.apr.2018.12.015.
[64] Chen J, Gong Y, Wang S, Guan B, Balkovic J, Kraxner F. To burn or retain crop residues on croplands? An integrated analysis of crop residue management in China. Sci Total Environ 2019;662:141–50. https://doi.org/10.1016/j.scitotenv.2019.01.150.
[65] Sarkar J, Mridha D, Sarkar J, Orasugh JT, Gangopadhyay B, Chattopadhyay D, et al. Synthesis of nanosilica from agricultural wastes and its multifaceted applications: a review. Biocatal Agric Biotechnol 2021;37:102175. https://doi.org/10.1016/j.bcab.2021.102175.
[66] Kirchherr J, Reike D, Hekkert M. Conceptualizing the circular economy: an analysis of 114 definitions. Resour Conserv Recycl 2017;127:221–32. https://doi.org/10.1016/j.resconrec.2017.09.005.
[67] Ghisellini P, Cialani C, Ulgiati S. A review on circular economy: the expected transition to a balanced interplay of environmental and economic systems. J Clean Prod 2016;114:11–32. https://doi.org/10.1016/j.jclepro.2015.09.007.
[68] Mittal S, Ahlgren EO, Shukla PR. Barriers to biogas dissemination in India: a review. Energy Pol 2018;112:361–70. https://doi.org/10.1016/j.enpol.2017.10.027.
[69] Chang I-S, Wu J, Zhou C, Shi M, Yang Y. A time-geographical approach to biogas potential analysis of China. Renew Sustain Energy Rev 2014;37:318–33. https://doi.org/10.1016/j.rser.2014.05.033.
[70] Igliński B, Piechota G, Iwański P, Skarzatek M, Pilarski G. 15 Years of the Polish agricultural biogas plants: their history, current status, biogas potential and perspectives. Clean Technol Environ Policy 2020;22:281–307. https://doi.org/10.1007/s10098-020-01812-3.
[71] Li Y, Park SY, Zhu J. Solid-state anaerobic digestion for methane production from organic waste. Renew Sustain Energy Rev 2011;15:821–6. https://doi.org/10.1016/j.rser.2010.07.042.
[72] Cheng F, Brewer CE. Conversion of protein-rich lignocellulosic wastes to bio-energy: review and recommendations for hydrolysis + fermentation and anaerobic digestion. Renew Sustain Energy Rev 2021;146:111167. https://doi.org/10.1016/j.rser.2021.111167.
[73] Taherdanak M, Zilouei H. Improving biogas production from wheat plant using alkaline pretreatment. Fuel 2014;115:714–9. https://doi.org/10.1016/j.fuel.2013.07.094.
[74] Angelidaki I, Karakashev D, Batstone DJ, Plugge CM, Stams AJM. Chapter sixteen - biomethanation and its potential. In: Rosenzweig AC, Ragsdale SW, editors. Methods enzymol. Academic Press; 2011. p. 327–51. https://doi.org/10.1016/B978-0-12-385112-3.00016-0.
[75] Ali S, Shah TA, Afzal A, Tabassum R. Exploring lignocellulosic biomass for bio-methane potential by anaerobic digestion and its economic feasibility. Energy Environ 2018;29:742–51. https://doi.org/10.1177/0958305X18759009.
[76] Izumi K, Okishio Y, Nagao N, Niwa C, Yamamoto S, Toda T. Effects of particle size on anaerobic digestion of food waste. Int Biodeterior Biodegrad 2010;64:601–8. https://doi.org/10.1016/j.ibiod.2010.06.013.
[77] Chen Y, Yang H, Zhao Z, Zou H, Zhu R, Jiang Q, et al. Comprehensively evaluating the digestive performance of sludge with different lignocellulosic components in mesophilic anaerobic digester. Bioresour Technol 2019;293:122042. https://doi.org/10.1016/j.biortech.2019.122042.
[78] Taifouris MR, Martín M. Multiscale scheme for the optimal use of residues for the production of biogas across Castile and Leon. J Clean Prod 2018;185:239–51. https://doi.org/10.1016/j.jclepro.2018.03.018.
[79] Wei S, Zhang H, Cai X, Xu J, Fang J, Liu H. Psychrophilic anaerobic co-digestion of highland barley straw with two animal manures at high altitude for enhancing biogas production. Energy Convers Manag 2014;88:40–8. https://doi.org/10.1016/j.enconman.2014.08.018.
[80] Krause MJ, Chickering GW, Townsend TG, Pullammanappallil P. 温度和粒径. Waste Manag 2018;71:25–30. https://doi.org/10.1016/j.wasman.2017.11.015.
[81] Guilford NGH, Lee HP, Kanger K, Meyer T, Edwards EA. Solid-state anaerobic digestion of mixed organic waste: the synergistic effect of food waste addition on the destruction of paper and cardboard. Environ Sci Technol 2019;53:12677–87. https://doi.org/10.1021/acs.est.9b04644.
[82] Mirmohamadsadeghi S, Karimi K, Azarbaijani R, Parsa Yeganeh L, Angelidaki I, Nizami A-S, et al. Pretreatment of lignocelluloses for enhanced biogas production: a review on influencing mechanisms and the importance of microbial diversity. Renew Sustain Energy Rev 2021;135:110173. https://doi.org/10.1016/j.rser.2020.110173.
[83] Pearse LF, Hettiaratchi JP, Kumar S. Towards developing a representative biochemical methane potential (BMP) assay for landfilled municipal solid waste – a review. Bioresour Technol 2018;254:312–24. https://doi.org/10.1016/j.biortech.2018.01.069.
[84] Barik D, Murugan S. Assessment of sustainable biogas production from de-oiled seed cake of karanja-an organic industrial waste from biodiesel industries. Fuel 2015;148:25–31. https://doi.org/10.1016/j.fuel.2015.01.072.
[85] Khalid A, Arshad M, Anjum M, Mahmood T, Dawson L. The anaerobic digestion of solid organic waste. Waste Manag 2011;31:1737–44. https://doi.org/10.1016/j.wasman.2011.03.021.
[86] Park Y-J, Tsuno H, Hidaka T, Cheon J-H. Evaluation of operational parameters in thermophilic acid fermentation of kitchen waste. J Mater Cycles Waste Manag 2008;10:46–52. https://doi.org/10.1007/s10163-007-0184-y.
[87] Lee DH, Behera SK, Kim JW, Park H-S. Methane production potential of leachate generated from Korean food waste recycling facilities: a lab-scale study. Waste Manag 2009;29:876–82. https://doi.org/10.1016/j.wasman.2008.06.033.
[88] Zhang C, Su H, Baeyens J, Tan T. Reviewing the anaerobic digestion of food waste for biogas production. Renew Sustain Energy Rev 2014;38:383–92. https://doi.org/10.1016/j.rser.2014.05.038.
[89] Bouallagui H, Ben Cheikh R, Marouani L, Hamdi M. Mesophilic biogas production from fruit and vegetable waste in a tubular digester. Bioresour Technol 2003;86:85–9. https://doi.org/10.1016/S0960-8524(02)00097-4.
[90] Yu HQ, Fang HHP. Acidogenesis of gelatin-rich wastewater in an upflow anaerobic reactor: influence of pH and temperature. Water Res 2003;37:55–66. https://doi.org/10.1016/S0043-1354(02)00256-7.
[91] Huber H, Thomm M, König H, Thies G, Stetter KO. Methanococcus thermolithotrophicus, a novel thermophilic lithotrophic methanogen. Arch Microbiol 1982;132:47–50. https://doi.org/10.1007/BF00690816.
[92] Matheri AN, Ntuli F, Ngila JC, Seodigeng T, Zvinowanda C, Njenga CK. Quantitative characterization of carbonaceous and lignocellulosic biomass for

anaerobic digestion. Renew Sustain Energy Rev 2018;92:9–16. https://doi.org/10.1016/j.rser.2018.04.070.
[93] Kondusamy D, Kalamdhad AS. Pre-treatment and anaerobic digestion of food waste for high rate methane production – a review. J Environ Chem Eng 2014;2:1821–30. https://doi.org/10.1016/j.jece.2014.07.024.
[94] Zhang L, Lee Y-W, Jahng D. Anaerobic co-digestion of food waste and piggery wastewater: focusing on the role of trace elements. Bioresour Technol 2011;102:5048–59. https://doi.org/10.1016/j.biortech.2011.01.082.
[95] Romero-Güiza MS, Vila J, Mata-Alvarez J, Chimenos JM, Astals S. The role of additives on anaerobic digestion: a review. Renew Sustain Energy Rev 2016;58:1486–99. https://doi.org/10.1016/j.rser.2015.12.094.
[96] Wang X, Barlaz MA. Decomposition and carbon storage of hardwood and softwood branches in laboratory-scale landfills. Sci Total Environ 2016;557–558:355–62. https://doi.org/10.1016/j.scitotenv.2016.03.091.
[97] Wang X, Padgett JM, Powell JS, Barlaz MA. Decomposition of forest products buried in landfills. Waste Manag 2013;33:2267–76. https://doi.org/10.1016/j.wasman.2013.07.009.
[98] Muaaz-Us-Salam S, Cleall PJ, Harbottle MJ. Application of enzymatic and bacterial biodelignification systems for enhanced breakdown of model lignocellulosic wastes. Sci Total Environ 2020;728:138741. https://doi.org/10.1016/j.scitotenv.2020.138741.
[99] Kumar S, Paritosh K, Pareek N, Chawade A, Vivekanand V. De-construction of major Indian cereal crop residues through chemical pretreatment for improved biogas production: an overview. Renew Sustain Energy Rev 2018;90:160–70. https://doi.org/10.1016/j.rser.2018.03.049.
[100] Liu X, Bayard R, Benbelkacem H, Buffière P, Gourdon R. Evaluation of the correlations between biodegradability of lignocellulosic feedstocks in anaerobic digestion process and their biochemical characteristics. Biomass Bioenergy 2015;81:534–43. https://doi.org/10.1016/j.biombioe.2015.06.021.
[101] Chen Y, Cheng JJ, Creamer KS. Inhibition of anaerobic digestion process: a review. Bioresour Technol 2008;99:4044–64. https://doi.org/10.1016/j.biortech.2007.01.057.
[102] Zhang Y, Huang M, Su J, Hu H, Yang M, Huang Z, et al. Overcoming biomass recalcitrance by synergistic pretreatment of mechanical activation and metal salt for enhancing enzymatic conversion of lignocellulose. Biotechnol Biofuels 2019;12:12. https://doi.org/10.1186/s13068-019-1354-6.
[103] Himmel ME, Ding S-Y, Johnson DK, Adney WS, Nimlos MR, Brady JW, et al. Biomass recalcitrance: engineering plants and enzymes for biofuels production. Science 2007;315:804–7. https://doi.org/10.1126/science.1137016.
[104] Grabber JH. How do lignin composition, structure, and cross-linking affect degradability? A review of cell wall model studies. Crop Sci 2005;45:820–31. https://doi.org/10.2135/cropsci2004.0191.
[105] Bugg TDH, Ahmad M, Hardiman EM, Rahmanpour R. Pathways for degradation of lignin in bacteria and fungi. Nat Prod Rep 2011;28:1883–96. https://doi.org/10.1039/C1NP00042J.
[106] Brandt A, Gräsvik J, Hallett J P, Welton T. Deconstruction of lignocellulosic biomass with ionic liquids. Green Chem 2013;15:550–83. https://doi.org/10.1039/C2GC36364J.
[107] Pan X, Xie D, Gilkes N, Gregg DJ, Saddler JN. Strategies to enhance the enzymatic hydrolysis of pretreated softwood with high residual lignin content. Appl Biochem Biotechnol 2005;124:1069. https://doi.org/10.1385/ABAB:124:1-3.
[108] Berlin A, Balakshin M, Gilkes N, Kadla J, Maximenko V, Kubo S, et al. Inhibition of cellulase, xylanase and β-glucosidase activities by softwood lignin preparations. J Biotechnol 2006;125:198–209. https://doi.org/10.1016/j.jbiotec.2006.02.021.
[109] Khan MU, Ahring BK. Lignin degradation under anaerobic digestion: influence of lignin modifications -A review. Biomass Bioenergy 2019;128:105325. https://doi.org/10.1016/j.biombioe.2019.105325.
[110] Sun D, Lv Z-W, Rao J, Tian R, Sun S-N, Peng F. Effects of hydrothermal pretreatment on the dissolution and structural evolution of hemicelluloses and lignin: a review. Carbohydr Polym 2022;281:119050. https://doi.org/10.1016/j.carbpol.2021.119050.
[111] Kalia S, Dufresne A, Cherian BM, Kaith BS, Avérous L, Njuguna J, et al. Cellulose-based bio- and nanocomposites: a review. Int J Polym Sci 2011;2011:e837875. https://doi.org/10.1155/2011/837875.
[112] Saha BC. Hemicellulose bioconversion. J Ind Microbiol Biotechnol 2003;30:279–91. https://doi.org/10.1007/s10295-003-0049-x.
[113] Fernández-Rodríguez J, Erdocia X, Sánchez C, González Alriols M, Labidi J. Lignin depolymerization for phenolic monomers production by sustainable processes. J Energy Chem 2017;26:622–31. https://doi.org/10.1016/j.jechem.2017.02.007.
[114] Gosselink RJA, Dam JEG van, Jong E de, Scott EL, Sanders JPM, Li J, et al. Fractionation, analysis, and PCA modeling of properties of four technical lignins for prediction of their application potential in. binders 2010;64:193–200. https://doi.org/10.1515/hf.2010.023.
[115] Harmsen PFH, Huijgen W, Bermudez L, Bakker R. Literature review of physical and chemical pretreatment processes for lignocellulosic biomass. Wageningen: Wageningen UR - Food & Biobased Research; 2010.
[116] Kim W, Kim Y-B, Yoon J, Roh H-J, Noh H-S. Method for promoting production of biogas using pancreatin in an anaerobic digestion process. US20150175462A1; 2015.
[117] Chen X, Gu Y, Zhou X, Zhang Y. Asparagus stem as a new lignocellulosic biomass feedstock for anaerobic digestion: increasing hydrolysis rate, methane production and biodegradability by alkaline pretreatment. Bioresour Technol 2014;164:78–85. https://doi.org/10.1016/j.biortech.2014.04.070.
[118] Gunes B, Stokes J, Davis P, Connolly C, Lawler J. Pre-treatments to enhance biogas yield and quality from anaerobic digestion of whiskey distillery and brewery wastes: a review. Renew Sustain Energy Rev 2019;113:109281. https://doi.org/10.1016/j.rser.2019.109281.
[119] Mu L, Zhang L, Ma J, Zhu K, Chen C, Li A. Enhancement of anaerobic digestion of phoenix tree leaf by mild alkali pretreatment: optimization by Taguchi orthogonal design and semi-continuous operation. Bioresour Technol 2020;313:123634. https://doi.org/10.1016/j.biortech.2020.123634.
[120] Yin C-R, Seo D-I, Kim M-K, Lee S-T. Inhibitory effect of hardwood lignin on acetate-utilizing methanogens in anaerobic digester sludge. Biotechnol Lett 2000;22:1531–5. https://doi.org/10.1023/A:1005620726731.
[121] Ozbayram EG, Ince O, Ince B, Harms H, Kleinsteuber S. Comparison of rumen and manure microbiomes and implications for the inoculation of anaerobic digesters. Microorganisms 2018;6:15. https://doi.org/10.3390/microorganisms6010015.
[122] Lu Q, Yu Z, Wang L, Liang Z, Li H, Sun L, et al. Sludge pre-treatments change performance and microbiome in methanogenic sludge digesters by releasing different sludge organic matter. Bioresour Technol 2020;316:123909. https://doi.org/10.1016/j.biortech.2020.123909.
[123] Bhatia SK, Kim S-H, Yoon J-J, Yang Y-H. Current status and strategies for second generation biofuel production using microbial systems. Energy Convers Manag 2017;148:1142–56. https://doi.org/10.1016/j.enconman.2017.06.073.
[124] Wan C, Li Y. Fungal pretreatment of lignocellulosic biomass. Biotechnol Adv 2012;30:1447–57. https://doi.org/10.1016/j.biotechadv.2012.03.003.
[125] Sharma HK, Xu C, Qin W. Biological pretreatment of lignocellulosic biomass for biofuels and bioproducts: an overview. Waste Biomass Valorization 2019;10:235–51. https://doi.org/10.1007/s12649-017-0059-y.
[126] Suhara H, Kodama S, Kamei I, Maekawa N, Meguro S. Screening of selective lignin-degrading basidiomycetes and biological pretreatment for enzymatic hydrolysis of bamboo culms. Int Biodeterior Biodegrad 2012;75:176–80. https://doi.org/10.1016/j.ibiod.2012.05.042.
[127] Cianchetta S, Di Maggio B, Burzi PL, Galletti S. Evaluation of selected white-rot fungal isolates for improving the sugar yield from wheat straw. Appl Biochem Biotechnol 2014;173:609–23. https://doi.org/10.1007/s12010-014-0869-3.
[128] Wan C, Li Y. Effectiveness of microbial pretreatment by Ceriporiopsis subvermispora on different biomass feedstocks. Bioresour Technol 2011;102:7507–12. https://doi.org/10.1016/j.biortech.2011.05.026.
[129] Díaz GV, Coniglio RO, Velazquez JE, Zapata PD, Villalba L, Fonseca MI. Adding value to lignocellulosic wastes via their use for endoxylanase production by Aspergillus fungi. Mycologia 2019;111:195–205. https://doi.org/10.1080/00275514.2018.1556557.
[130] Baghbanzadeh M, Savage J, Balde H, Sartaj M, VanderZaag AC, Abdehagh N, et al. Enhancing hydrolysis and bio-methane generation of extruded lignocellulosic wood waste using microbial pre-treatment. Renew Energy 2021;170:438–48. https://doi.org/10.1016/j.renene.2021.01.131.
[131] Rashid Gm m, Durán-Peña Mj, Rahmanpour R, Sapsford D, Bugg T d h. Delignification and enhanced gas release from soil containing lignocellulose by treatment with bacterial lignin degraders. J Appl Microbiol 2017;123:159–71. https://doi.org/10.1111/jam.13470.
[132] Ahmad M, Roberts JN, Hardiman EM, Singh R, Eltis LD, Bugg TDH. Identification of DypB from Rhodococcus jostii RHA1 as a lignin peroxidase. Biochemistry 2011;50:5096–107. https://doi.org/10.1021/bi101892z.
[133] Wieczorek N, Kucuker MA, Kuchta K. Fermentative hydrogen and methane production from microalgal biomass (Chlorella vulgaris) in a two-stage combined process. Appl Energy 2014;132:108–17. https://doi.org/10.1016/j.apenergy.2014.07.003.
[134] Bu J, Tian Q-Q, Zhu M-J. Enhanced biodegradation of sugar cane bagasse by coculture of Clostridium thermocellum and Thermoanaerobacterium aotearoense supplemented with CaCO3. Energy Fuel 2017;31:9477–83. https://doi.org/10.1021/acs.energyfuels.7b01362.
[135] Karuppiah V, Zhixiang L, Liu H, Vallikkannu M, Chen J. Co-culture of Vel1-overexpressed Trichoderma asperellum and Bacillus amyloliquefaciens: an eco-friendly strategy to hydrolyze the lignocellulose biomass in soil to enrich the soil fertility, plant growth and disease resistance. Microb Cell Factories 2021;20:57. https://doi.org/10.1186/s12934-021-01540-3.
[136] Taha M, Shahsavari E, Al-Hothaly K, Mouradov A, Smith AT, Ball AS, et al. Enhanced biological straw saccharification through coculturing of lignocellulose-degrading microorganisms. Appl Biochem Biotechnol 2015;175:3709–28. https://doi.org/10.1007/s12010-015-1539-9.
[137] Zhang F, Zhao X, Bai F. Improvement of cellulase production in Trichoderma reesei Rut-C30 by overexpression of a novel regulatory gene Trvib-1. Bioresour Technol 2018;247:676–83. https://doi.org/10.1016/j.biortech.2017.09.126.
[138] Lin L, Wang S, Li X, He Q, Benz JP, Tian C. STK-12 acts as a transcriptional brake to control the expression of cellulase-encoding genes in Neurospora crassa. PLoS Genet 2019;15:e1008510. https://doi.org/10.1371/journal.pgen.1008510.
[139] Ko JK, Enkh-Amgalan T, Gong G, Um Y, Lee S-M. Improved bioconversion of lignocellulosic biomass by Saccharomyces cerevisiae engineered for tolerance to acetic acid. GCB Bioenergy 2020;12:90–100. https://doi.org/10.1111/gcbb.12656.
[140] You Z, Zhang S, Kim H, Chiang P-C, Sun Y, Guo Z, et al. Effects of corn stover pretreated with NaOH and CaO on anaerobic Co-digestion of swine manure and corn stover. Appl Sci 2019;9:123. https://doi.org/10.3390/app9010123.
[141] Eduok S, John O, Ita B, Inyang E, Coulon F. Enhanced biogas production from anaerobic Co-digestion of lignocellulosic biomass and poultry feces using source separated human urine as buffering agent. Front Environ Sci 2018;6.
[142] David A, Govil T, Tripathi AK, McGeary J, Farrar K, Sani RK. Thermophilic anaerobic digestion: enhanced and sustainable methane production from Co-digestion of food and lignocellulosic wastes. Energies 2018;11:2058. https://doi.org/10.3390/en11082058.

[143] Meraj S, Liaquat R, Raza Naqvi S, Sheikh Z, Zainab A, Khoja AH, et al. Enhanced methane production from anaerobic Co-digestion of wheat straw rice straw and sugarcane bagasse: a kinetic analysis. Appl Sci 2021;11:6069. https://doi.org/10.3390/app11136069.
[144] Cong W-F, Moset V, Feng L, Møller HB, Eriksen J. Anaerobic co-digestion of grass and forbs – influence of cattle manure or grass based inoculum. Biomass Bioenergy 2018;119:90–6. https://doi.org/10.1016/j.biombioe.2018.09.009.
[145] Meneses-Quelal WO, Velázquez-Martí B, Gaibor-Chávez J, Niño-Ruiz Z, Ferrer-Gisbert A. Anaerobic Co-digestion of slaughter residues with agricultural waste of amaranth quinoa and wheat. BioEnergy Res 2021. https://doi.org/10.1007/s12155-021-10350-9.
[146] Zou H, Chen Y, Shi J, Zhao T, Yu Q, Yu S, et al. Mesophilic anaerobic co-digestion of residual sludge with different lignocellulosic wastes in the batch digester. Bioresour Technol 2018;268:371–81. https://doi.org/10.1016/j.biortech.2018.07.129.
[147] Simioni T, Agustini CB, Dettmer A, Gutterres M. Anaerobic Co-digestion of tannery wastes and untreated/pretreated oat straw. BioEnergy Res 2022;15:589–601. https://doi.org/10.1007/s12155-021-10285-1.
[148] Nkemka VN, Gilroyed B, Yanke J, Gruninger R, Vedres D, McAllister T, et al. Bioaugmentation with an anaerobic fungus in a two-stage process for biohydrogen and biogas production using corn silage and cattail. Bioresour Technol 2015;185:79–88. https://doi.org/10.1016/j.biortech.2015.02.100.
[149] Kurt Kara G, Doluk R, Civelek Yoruklu H, Demir A, Ozkaya B. Biomethane production kinetics of rumen pretreated lignocellulosic wastes. Clean Technol Environ Policy 2021;23:2941–54. https://doi.org/10.1007/s10098-021-02214-9.
[150] Singh nee' Nigam P, Gupta N, Anthwal A. Pre-treatment of agro-industrial residues. In: Singh nee' Nigam P, Pandey A, editors. Biotechnol. Agro-ind. Residues util. Util. Agro-residues. Dordrecht: Springer Netherlands; 2009. p. 13–33. https://doi.org/10.1007/978-1-4020-9942-7_2.
[151] Niladevi KN. Ligninolytic enzymes. In: Singh nee' Nigam P, Pandey A, editors. Biotechnol. Agro-ind. Residues util. Util. Agro-residues. Dordrecht: Springer Netherlands; 2009. p. 397–414. https://doi.org/10.1007/978-1-4020-9942-7_22.
[152] Ahmad M, Hardiman EM, Singh R. The emerging role for bacteria in lignin degradation and bio-product formation. Curr Opin Biotechnol 2011;22:394–400. https://doi.org/10.1016/j.copbio.2010.10.009.
[153] Horn SJ, Vaaje-Kolstad G, Westereng B, Eijsink V. Novel enzymes for the degradation of cellulose. Biotechnol Biofuels 2012;5:45. https://doi.org/10.1186/1754-6834-5-45.
[154] Siqueira JGW, Rodrigues C, Vandenberghe LP de S, Woiciechowski AL, Soccol CR. Current advances in on-site cellulase production and application on lignocellulosic biomass conversion to biofuels: a review. Biomass Bioenergy 2020;132:105419. https://doi.org/10.1016/j.biombioe.2019.105419.
[155] Pérez J, Muñoz-Dorado J, de la Rubia T, Martínez J. Biodegradation and biological treatments of cellulose, hemicellulose and lignin: an overview. Int Microbiol 2002;5:53–63. https://doi.org/10.1007/s10123-002-0062-3.
[156] Eibinger M, Ganner T, Bubner P, Rošker S, Kracher D, Haltrich D, et al. Cellulose surface degradation by a lytic polysaccharide monooxygenase and its effect on cellulase hydrolytic efficiency. J Biol Chem 2014;289:35929–38. https://doi.org/10.1074/jbc.M114.602227.
[157] Singh SK. Biological treatment of plant biomass and factors affecting bioactivity. J Clean Prod 2021;279:123546. https://doi.org/10.1016/j.jclepro.2020.123546.
[158] Schimpf U, Hanreich A, Mähnert P, Unmack T, Junne S, Renpenning J, et al. Improving the efficiency of large-scale biogas processes. In: Pectinolytic enzymes accelerate the lignocellulose degradation. vol. 8; 2013.
[159] Parmar N, Singh A, Ward OP. Enzyme treatment to reduce solids and improve settling of sewage sludge. J Ind Microbiol Biotechnol 2001;26:383–6. https://doi.org/10.1038/sj.jim.7000150.
[160] Sánchez C. Lignocellulosic residues: biodegradation and bioconversion by fungi. Biotechnol Adv 2009;27:185–94. https://doi.org/10.1016/j.biotechadv.2008.11.001.
[161] Arantes V, Milagres AMF. The synergistic action of ligninolytic enzymes (MnP and Laccase) and Fe3+-reducing activity from white-rot fungi for degradation of Azure B. Enzym Microb Technol 2007;42:17–22. https://doi.org/10.1016/j.enzmictec.2007.07.017.
[162] Shary S, Kapich AN, Panisko EA, Magnuson JK, Cullen D, Hammel KE. Differential expression in Phanerochaete chrysosporium of membrane-associated proteins relevant to lignin degradation. Appl Environ Microbiol 2008;74:7252–7. https://doi.org/10.1128/AEM.01997-08.
[163] Ljungdahl LG. The cellulase/hemicellulase system of the anaerobic FungusOrpinomycesPC-2 and aspects of its applied use. Ann N Y Acad Sci 2008;1125:308–21. https://doi.org/10.1196/annals.1419.030.
[164] Zhang H, Wang X. Modular co-culture engineering, a new approach for metabolic engineering. Metab Eng 2016;37:114–21. https://doi.org/10.1016/j.ymben.2016.05.007.
[165] Abdel-Hamid AM, Solbiati JO, Cann IKO. Chapter one - insights into lignin degradation and its potential industrial applications. In: Sariaslani S, Gadd GM, editors. Adv. Appl. Microbiol. Academic Press; 2013. p. 1–28. https://doi.org/10.1016/B978-0-12-407679-2.00001-6.
[166] Monrroy M, Ortega I, Ramírez M, Baeza J, Freer J. Structural change in wood by brown rot fungi and effect on enzymatic hydrolysis. Enzym Microb Technol 2011;49:472–7. https://doi.org/10.1016/j.enzmictec.2011.08.004.
[167] Ahmad M, Taylor CR, Pink D, Burton K, Eastwood D, Bending GD, et al. Development of novel assays for lignin degradation: comparative analysis of bacterial and fungal lignin degraders. Mol Biosyst 2010;6:815–21. https://doi.org/10.1039/B908966G.
[168] Zimmermann W. Degradation of lignin by bacteria. J Biotechnol 1990;13:119–30. https://doi.org/10.1016/0168-1656(90)90098-V.
[169] Taylor Cr, Hardiman Em, Ahmad M, Sainsbury Pd, Norris Pr, Bugg Td h. Isolation of bacterial strains able to metabolize lignin from screening of environmental samples. J Appl Microbiol 2012;113:521–30. https://doi.org/10.1111/j.1365-2672.2012.05352.x.
[170] Bugg TDH, Williamson JJ, Rashid GMM. Bacterial enzymes for lignin depolymerisation: new biocatalysts for generation of renewable chemicals from biomass. Curr Opin Chem Biol 2020;55:26–33. https://doi.org/10.1016/j.cbpa.2019.11.007.
[171] Granja-Travez RS, Persinoti GF, Squina FM, Bugg TDH. Functional genomic analysis of bacterial lignin degraders: diversity in mechanisms of lignin oxidation and metabolism. Appl Microbiol Biotechnol 2020;104:3305–20. https://doi.org/10.1007/s00253-019-10318-y.
[172] Paudel YP, Qin W. Characterization of novel cellulase-producing bacteria isolated from rotting wood samples. Appl Biochem Biotechnol 2015;177:1186–98. https://doi.org/10.1007/s12010-015-1806-9.
[173] Rico JL, Reardon KF, De Long SK. Inoculum microbiome composition impacts fatty acid product profile from cellulosic feedstock. Bioresour Technol 2021;323:124532. https://doi.org/10.1016/j.biortech.2020.124532.
[174] Kong X, Du J, Ye X, Xi Y, Jin H, Zhang M, et al. Enhanced methane production from wheat straw with the assistance of lignocellulolytic microbial consortium TC-5. Bioresour Technol 2018;263:33–9. https://doi.org/10.1016/j.biortech.2018.04.079.
[175] Jiang Y, Zhang T, Lu J, Dürre P, Zhang W, Dong W, et al. Microbial co-culturing systems: butanol production from organic wastes through consolidated bioprocessing. Appl Microbiol Biotechnol 2018;102:5419–25. https://doi.org/10.1007/s00253-018-8970-0.
[176] Wu G, Yan Q, Jones JA, Tang YJ, Fong SS, Koffas MAG. Metabolic burden: cornerstones in synthetic biology and metabolic engineering applications. Trends Biotechnol 2016;34:652–64. https://doi.org/10.1016/j.tibtech.2016.02.010.
[177] Zhou K, Qiao K, Edgar S, Stephanopoulos G. Distributing a metabolic pathway among a microbial consortium enhances production of natural products. Nat Biotechnol 2015;33:377–83. https://doi.org/10.1038/nbt.3095.
[178] Vyu Artsatbanov, Vostroknutova GN, Shleeva MO, Goncharenko AV, Zinin AI, Ostrovsky DN, et al. Influence of oxidative and nitrosative stress on accumulation of diphosphate intermediates of the non-mevalonate pathway of isoprenoid biosynthesis in corynebacteria and mycobacteria. Biochem Mosc 2012;77:362–71. https://doi.org/10.1134/S0006297912040074.
[179] Kato S, Haruta S, Cui ZJ, Ishii M, Igarashi Y. Effective cellulose degradation by a mixed-culture system composed of a cellulolytic Clostridium and aerobic non-cellulolytic bacteria. FEMS Microbiol Ecol 2004;51:133–42. https://doi.org/10.1016/j.femsec.2004.07.015.
[180] Chandra R, Raj A, Purohit HJ, Kapley A. Characterisation and optimisation of three potential aerobic bacterial strains for kraft lignin degradation from pulp paper waste. Chemosphere 2007;67:839–46. https://doi.org/10.1016/j.chemosphere.2006.10.011.
[181] Weimer PJ, Nerdahl M, Brandl DJ. Production of medium-chain volatile fatty acids by mixed ruminal microorganisms is enhanced by ethanol in co-culture with Clostridium kluyveri. Bioresour Technol 2015;175:97–101. https://doi.org/10.1016/j.biortech.2014.10.054.
[182] Kato S, Haruta S, Cui ZJ, Ishii M, Igarashi Y. Network relationships of bacteria in a stable mixed culture. Microb Ecol 2008;56:403–11. https://doi.org/10.1007/s00248-007-9357-4.
[183] Wen Z, Liao W, Chen S. Production of cellulase/β-glucosidase by the mixed fungi culture Trichoderma reesei and Aspergillus phoenicis on dairy manure. Process Biochem 2005;40:3087–94. https://doi.org/10.1016/j.procbio.2005.03.044.
[184] Chi Y, Hatakka A, Maijala P. Can co-culturing of two white-rot fungi increase lignin degradation and the production of lignin-degrading enzymes? Int Biodeterior Biodegrad 2007;59:32–9. https://doi.org/10.1016/j.ibiod.2006.06.025.
[185] Cohan FM, Koeppel AF. The origins of ecological diversity in prokaryotes. Curr Biol 2008;18:R1024–34. https://doi.org/10.1016/j.cub.2008.09.014.
[186] Lu H, Yadav V, Zhong M, Bilal M, Taherzadeh MJ, Iqbal HMN. Bioengineered microbial platforms for biomass-derived biofuel production – a review. Chemosphere 2022;288:132528. https://doi.org/10.1016/j.chemosphere.2021.132528.
[187] Tabatabaei M, Aghbashlo M, Valijanian E, Kazemi Shariat Panahi H, Nizami A-S, Ghanavati H, et al. A comprehensive review on recent biological innovations to improve biogas production, Part 1: upstream strategies. Renew Energy 2020;146:1204–20. https://doi.org/10.1016/j.renene.2019.07.037.
[188] Majidian P, Tabatabaei M, Zeinolabedini M, Naghshbandi MP, Chisti Y. Metabolic engineering of microorganisms for biofuel production. Renew Sustain Energy Rev 2018;82:3863–85. https://doi.org/10.1016/j.rser.2017.10.085.
[189] Rollin JA, Martin del Campo J, Myung S, Sun F, You C, Bakovic A, et al. High-yield hydrogen production from biomass by in vitro metabolic engineering: mixed sugars coutilization and kinetic modeling. Proc Natl Acad Sci USA 2015;112:4964–9. https://doi.org/10.1073/pnas.1417719112.
[190] Jeffries TW, Jin Y-S. Metabolic engineering for improved fermentation of pentoses by yeasts. Appl Microbiol Biotechnol 2004;63:495–509. https://doi.org/10.1007/s00253-003-1450-0.
[191] Sedlak M, Edenberg HJ, Ho NWY. DNA microarray analysis of the expression of the genes encoding the major enzymes in ethanol production during glucose and xylose co-fermentation by metabolically engineered Saccharomyces yeast. Enzym Microb Technol 2003;33:19–28. https://doi.org/10.1016/S0141-0229(03)00067-X.

[192] Divya D, Gopinath LR, Merlin Christy P. A review on current aspects and diverse prospects for enhancing biogas production in sustainable means. Renew Sustain Energy Rev 2015;42:690–9. https://doi.org/10.1016/j.rser.2014.10.055.
[193] Azman S, Khadem AF, van Lier JB, Zeeman G, Plugge CM. Presence and role of anaerobic hydrolytic microbes in conversion of lignocellulosic biomass for biogas production. Crit Rev Environ Sci Technol 2015;45:2523–64. https://doi.org/10.1080/10643389.2015.1053727.
[194] Tyagi VK, Fdez-Güelfo LA, Zhou Y, Álvarez-Gallego CJ, Garcia LIR, Ng WJ. Anaerobic co-digestion of organic fraction of municipal solid waste (OFMSW): progress and challenges. Renew Sustain Energy Rev 2018;93:380–99. https://doi.org/10.1016/j.rser.2018.05.051.
[195] Chiumenti A, da Borso F, Limina S. Dry anaerobic digestion of cow manure and agricultural products in a full-scale plant: efficiency and comparison with wet fermentation. Waste Manag 2018;71:704–10. https://doi.org/10.1016/j.wasman.2017.03.046.
[196] Alvarez R, Lidén G. Anaerobic co-digestion of aquatic flora and quinoa with manures from Bolivian Altiplano. Waste Manag 2008;28:1933–40. https://doi.org/10.1016/j.wasman.2007.11.002.
[197] Wang X, Yang G, Feng Y, Ren G, Han X. Optimizing feeding composition and carbon–nitrogen ratios for improved methane yield during anaerobic co-digestion of dairy, chicken manure and wheat straw. Bioresour Technol 2012;120:78–83. https://doi.org/10.1016/j.biortech.2012.06.058.
[198] Li D, Liu S, Mi L, Li Z, Yuan Y, Yan Z, et al. Effects of feedstock ratio and organic loading rate on the anaerobic mesophilic co-digestion of rice straw and cow manure. Bioresour Technol 2015;189:319–26. https://doi.org/10.1016/j.biortech.2015.04.033.
[199] Pagés-Díaz J, Pereda-Reyes I, Taherzadeh MJ, Sárvári-Horváth I, Lundin M. Anaerobic co-digestion of solid slaughterhouse wastes with agro-residues: synergistic and antagonistic interactions determined in batch digestion assays. Chem Eng J 2014;245:89–98. https://doi.org/10.1016/j.cej.2014.02.008.
[200] Neshat SA, Mohammadi M, Najafpour GD, Lahijani P. Anaerobic co-digestion of animal manures and lignocellulosic residues as a potent approach for sustainable biogas production. Renew Sustain Energy Rev 2017;79:308–22. https://doi.org/10.1016/j.rser.2017.05.137.
[201] Cheng X-Y, Zhong C. Effects of feed to inoculum ratio, Co-digestion, and pretreatment on biogas production from anaerobic digestion of cotton stalk. Energy Fuel 2014;28:3157–66. https://doi.org/10.1021/ef402562z.
[202] Shrestha S, Fonoll X, Khanal SK, Raskin L. Biological strategies for enhanced hydrolysis of lignocellulosic biomass during anaerobic digestion: current status and future perspectives. Bioresour Technol 2017;245:1245–57. https://doi.org/10.1016/j.biortech.2017.08.089.
[203] Brune A. Symbiotic digestion of lignocellulose in termite guts. Nat Rev Microbiol 2014;12:168–80. https://doi.org/10.1038/nrmicro3182.
[204] Geib SM, Filley TR, Hatcher PG, Hoover K, Carlson JE, Jimenez-Gasco M del M, et al. Lignin degradation in wood-feeding insects. Proc Natl Acad Sci USA 2008;105:12932–7. https://doi.org/10.1073/pnas.0805257105.
[205] Tartar A, Wheeler MM, Zhou X, Coy MR, Boucias DG, Scharf ME. Parallel metatranscriptome analyses of host and symbiont gene expression in the gut of the termite Reticulitermes flavipes. Biotechnol Biofuels 2009;2:25. https://doi.org/10.1186/1754-6834-2-25.
[206] Liang J, Nabi M, Zhang P, Zhang G, Cai Y, Wang Q, et al. Promising biological conversion of lignocellulosic biomass to renewable energy with rumen microorganisms: a comprehensive review. Renew Sustain Energy Rev 2020;134:110335. https://doi.org/10.1016/j.rser.2020.110335.
[207] Wall DM, Allen E, O'Shea R, O'Kiely P, Murphy JD. Investigating two-phase digestion of grass silage for demand-driven biogas applications: effect of particle size and rumen fluid addition. Renew Energy 2016;86:1215–23. https://doi.org/10.1016/j.renene.2015.09.049.
[208] Lavrenčič A, Pirman T. In vitro gas and short-chain fatty acid production from soybean meal treated with chestnut and quebracho wood extracts by using sheep rumen fluid. J Anim Feed Sci 2021;30:312–9. https://doi.org/10.22358/jafs/144587/2021.
[209] Carrillo-Barragán P, Dolfing J, Sallis P, Gray N. The stability of ethanol production from organic waste by a mixed culture depends on inoculum transfer time. Biochem Eng J 2021;166:107875. https://doi.org/10.1016/j.bej.2020.107875.
[210] Charles W, Walker L, Cord-Ruwisch R. Effect of pre-aeration and inoculum on the start-up of batch thermophilic anaerobic digestion of municipal solid waste. Bioresour Technol 2009;100:2329–35. https://doi.org/10.1016/j.biortech.2008.11.051.
[211] Botheju D, Lie B, Bakke R. Oxygen effects in anaerobic. Digestion 2009;30:191. https://doi.org/10.4173/mic.2009.4.1. –201.
[212] Jang HM, Park SK, Ha JH, Park JM. Microbial community structure in a thermophilic aerobic digester used as a sludge pretreatment process for the mesophilic anaerobic digestion and the enhancement of methane production. Bioresour Technol 2013;145:80–9. https://doi.org/10.1016/j.biortech.2013.01.094.
[213] Gil A. Current insights into lignocellulose related waste valorization. Chem Eng J Adv 2021;8:100186. https://doi.org/10.1016/j.ceja.2021.100186.
[214] NNFCC. Use of sustainably-sourced residue and waste streams for advanced biofuel production in the EU, https://www.nnfcc.co.uk/report-sustainable-residue-waste-advanced-biofuel (accessed September 8, 2022).
[215] Pathak BS, Chandel AK. Feedstock transportation, agricultural processing, logistic from farm to bio-refinery: recent developments, mechanization, and cost analysis. In: Chandel AK, Sukumaran RK, editors. Sustain. Biofuels dev. India. Cham: Springer International Publishing; 2017. p. 207–21. https://doi.org/10.1007/978-3-319-50219-9_9.
[216] NNFCC. Lignocellulosic feedstock in the UK, https://www.nnfcc.co.uk/reports-lignocellulosic-feedstock-uk (accessed September 8, 2022).
[217] Monlau F, Sambusiti C, Antoniou N, Barakat A, Zabaniotou A. A new concept for enhancing energy recovery from agricultural residues by coupling anaerobic digestion and pyrolysis process. Appl Energy 2015;148:32–8. https://doi.org/10.1016/j.apenergy.2015.03.024.
[218] Noorollahi Y, Kheirrouz M, Asl HF, Yousefi H, Hajinezhad A. Biogas production potential from livestock manure in Iran. Renew Sustain Energy Rev 2015;50:748–54. https://doi.org/10.1016/j.rser.2015.04.190.
[219] Joint Research Centre (European Commission), Agostini A, Marelli L, Edwards R, Giuntoli J. Solid and gaseous bioenergy pathways: input values and GHG emissions : calculated according to the methodology set in COM. LU: Publications Office of the European Union; 2016. p. 767.
[220] Koornneef J, van Breevoort P, Noothout P, Hendriks C, uchien Luning, Camps A. Global potential for biomethane production with carbon capture, transport and storage up to 2050. Energy Proc 2013;37:6043–52. https://doi.org/10.1016/j.egypro.2013.06.533.
[221] IRENA – International Renewable Energy Agency. Bioenergy, https://www.irena.org/bioenergy (accessed February 24, 2022).