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Review

Anaerobic Digestion of Food Waste—A Short Review

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Abstract: In recent years, growing environmental awareness, the need to reduce greenhouse gas emissions, and the energy crisis have led many countries to seek alternative energy sources. One of the most promising solutions is biogas production via anaerobic digestion (AD), whose substrate can be organic-rich and easily biodegradable food waste (FW). This waste is a significant part of the global waste problem, and its use for energy production is beneficial to both the environment and the economy. This paper presents important issues concerning the monitoring of the AD process, as well as standard and innovative, for the implementation of this process, technological solutions. The aim of the measures taken to optimise the process is to increase AD efficiency and obtain the highest possible methane content in biogas. Two approaches—pretreatment and anaerobic co-digestion (AcoD)—have been integral to the implementation of AD of food waste for years. They are presented in this paper based on a review of recent research developments. Pretreatment methods are discussed with particular emphasis on mechanical, chemical and biological methods. The AcoD of FW with different organic substrates has been extensively reviewed, as confirmed by numerous studies, where higher buffer capacity and optimum nutrient balance enhance the biogas/methane yields. Attention was also paid to the parameters, operating mode and configurations of anaerobic digesters, with a thorough analysis of the advantages and disadvantages of each solution. The article concludes with a brief presentation of the development perspectives for the discussed FW management method and recommendations.

Keywords: food waste; anaerobic digestion; pretreatment; bioreactor configurations; process efficiency

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1. Introduction

Faced with an energy crisis and climate change, the world is looking for green energy sources to replace fossil fuels. The security of energy supplies, especially renewable energy, and reduction of CO_2 emissions have become priorities in the energy and environmental policies developed by Poland and the EU. Anaerobic digestion (AD), a long-known microbial process for producing biogas, including methane, is currently seen as an alternative energy source [1–3]. In an effort to solve another civilisational problem, which is the high production of waste, organic waste, including food waste, is increasingly used as a substrate in the AD process [4–6].

Each year, one third of the world's food production is wasted in various ways, causing enormous social, environmental and economic problems [7–9]. The United Nations Food and Agriculture Organization (FAO) reports that 1 billion 300 million tons of food are wasted worldwide each year, which accounts for $^1/_3$ of all the food produced in the world. Moreover, it is believed that $^2/_3$ of the food that is thrown away is actually still fit for consumption. The total amount of wasted food at the EU level in 2020 was nearly 57 million tons of fresh weight. Of this, more than 31 million tons is food wasted in households, which accounts for 55 percent of the total amount. Processing and manufacturing (18 percent), where the amount of food waste measured was just over 10 million tons of fresh weight, took second place [10,11]. Recovering energy and nutrients from food waste is an opportunity to maintain a stable economy in countries around the world. Taking into account the negative impact of landfilling, incineration or composting of waste on the natural environment, anaerobic digestion appears to be the most appropriate and promising method of disposing of food waste, which, at the same time, is a source of biofuel—methane [12,13]. It is worth noting that the method of anaerobic biodegradation of waste is perfectly in line with the Sustainable Development Goals 2030 [14], which are further reflected in the priorities and strategies of the European Commission [15]. In the fields of energy and climate, the European Union is setting ambitious goals for 2030, which relate to, i.a., improving energy efficiency and increasing the share of energy from renewable sources. Within the framework of goals that include the concept of a closed (circular) economy [15], the Commission pledges to conserve resources and minimise waste, bearing in mind the context of the accelerating depletion of global resources. Such measures are supported by the development of investments in the construction of biogas facilities, which, globally, is expected to ensure competitiveness, enable the creation of jobs at the local level, bring social integration and provide encouragement for ecological innovation.

Food waste is an extremely promising substrate for the biogas generation process. Due to its high moisture content, it is an easily biodegradable material [16–18], while the organic matter it contains (about 90% VS, VS—volatile solids) makes it a valuable medium for bacteria. It is distinguished among other organic wastes by its high biochemical potential, BMP (200–670 $m^3 \cdot Mg^{-1}$ s.s.o) [4,19,20]. According to the literature, the AD process is most often carried out on waste food from restaurants [16–18,21], which is used on its own or in combination with other co-substrates. Laboratory testing of BMP is conducted with industrial production waste, including but not limited to sugar beet pulp, molasses, whey, fat, confectionery waste, waste fruits, vegetables, or even coffee [22–26], with the aim of using them—either on their own or in combination with co-substrates—on the technical scale.

The choice of biogas production technology depends on multiple factors. The type of raw material is the most important of these. Next comes economic viability, taking into account the capital expenditure incurred. The choice of raw material should be guided by the distance from the biogas plant itself, which later translates into the economics of the digestion process. Substrates from a nearby manufacturing plant create the most stable source of supply for a biogas plant, which is also the most advantageous from economic and logistical points of view [27,28]. The construction of biogas plants "at the source" and the use of waste biomass from the agri-food industry is the most favorable solution in the era of the climate and energy crisis. On the one hand, it closes the biogen loop in accordance with the principles of circular economy, and on the other hand, it serves as a source of electricity and heat. It is worth mentioning that biogas is a smart complement to other types of renewable energy sources, especially those dependent on variable weather conditions. It is a fitting solution for climate zones with limited capacity to capture solar energy.

Despite the numerous studies conducted with the view of optimising anaerobic digestion of food waste, the authors of recently published review papers [2,10,16,29] emphasise that in practice, there are still key technical problems preventing full implementation of the AD process as a method of utilising food waste. First and foremost among them is the low stability of the system, caused by the accumulation of volatile fatty acids, which is most often manifested by sudden drops in pH [30]. It should be mentioned here that the optimal range of pH values for methanogenic bacteria to function is between 6.5 and 7.2 [31]. The second factor holding back the development of anaerobic digestion of food waste is the low efficiency of biofermenters due to limitations in the organic loading rate (OLR). This is due to the chemical composition of food products, characterised by a high content of fats and carbohydrates, which, on the one hand, are significant carriers of energy but on the other hand, give an unfavourable boost to the dynamics of the decomposition of matter in the first phase of the process (hydrolysis). Recent years have brought an intensive development of research on various strategies to increase the share of anaerobic digestion in food waste management. Research centres around the world have been conducting research on the verification of the properties and biogas yield of food waste, selection of co-substrates, monitoring of key process parameters, and design of innovative biofermenters and the impact of process-enhancing additives (usually enzymes, carriers, micronutrients, or alkalising compounds) [32–36].

The aim of this paper is to present and analyse the most important issues concerning the production of biogas/methane from food waste based on the latest knowledge in this field and the latest research and scientific discoveries. The paper demonstrates the prospects for the development of anaerobic digestion as a method of utilising this type of waste and, above all, as a means of generating electricity and heat in times of energy crisis and economic change. This piece is a recommendation for improving and implementing anaerobic digestion of food waste, with specific suggestions.

2. Key Parameters

Regardless of the choice of substrate type, it is necessary to constantly monitor the parameters of anaerobic digestion in the digester. The key parameters affecting the process of anaerobic digestion include temperature, pH value, volatile fatty acid (VFA) concentrations, C/N ratio, organic loading rate (OLR), and hydraulic retention time (HRT). Anaerobic digestion is an absolutely anaerobic process. Some of the methanogenic bacteria die even in the presence of a small amount of oxygen [31]. The statement that a higher ambient temperature makes any chemical reaction proceed faster does not apply to biological processes. In the case of anaerobic digestion, it is important to consider the live microorganisms that determine the success of the entire process. Each type of bacteria involved in metabolic processes needs a different temperature. The same applies to pH and other parameters mentioned above, which are closely correlated with one another, while their monitoring contributes to the success of the anaerobic digestion process.

2.1. Temperature

Temperature is important in the biogas production process. The impact of temperature on the viscosity of the contents of the bioreactor and on the activity of microorganisms during the biogas generation process is well known. The AD process can occur under various temperature conditions. A distinction is made between psychrophilic (below 20 °C), mesophilic (25–40 °C) and thermophilic (45–60 °C) conditions [37,38]. Psychrophilic conditions are not popular in research into the AD of food waste, as the process carried out under these conditions requires a longer digestion time and has lower CH_4 (63.1–66.8%) and CO_2 yields (27.4–33.3%), while its biochemical methane potential (BMP) is 250–460 mL·g $^{-1}$ VS [39]. According to Muñoz et al. (2020), the advantage of psychrophilic conditions is that the bacteria have a higher tolerance to inhibitors compared to mesophilic and thermophilic conditions, which results in effective methane production [40]. The technology adapted to these conditions has been improved in recent years. Anaerobic membrane bioreactors (AnMBR) [41] and bioelectrochemical reactors [42] have been developed.

Both mesophilic and thermophilic conditions are popular in research into AD of food waste. It has been suggested that carrying out the process under mesophilic conditions is the most popular technique implemented on a technical scale, although mesophilic bacteria display a lower degree of pathogen inactivation and a lower degree of methane production compared to thermophilic bacteria. However, it should be taken into account that food waste contains a large amount of organic matter, which is associated with high values of organic loading rate (OLR), and mesophilic conditions make it possible to maintain higher OLR values in bioreactors than under thermophilic conditions [43]. Energy requirements for intermediate temperatures are also lower than for thermophilic conditions. The issues mentioned are related to the principle of technological restraint. Of course, the AD process carried out in the thermophilic range has numerous advantages: higher growth rate of methanogens [44], higher bioconversion rate [45], more efficient destruction of pathogens, shortened retention time and lower sensitivity to inhibitors, including ammonia. However, economic factors dictate the operation of biogas plants, including, in particular, agricultural biogas plants—under mesophilic conditions [46].

2.2. pH

The pH value is a key parameter for ensuring fermenter stability. During the anaerobic digestion process, the optimal pH value should be in the pH = 7 range. The anaerobic digestion process involves three types of bacteria: hydrolytic, fermentation and methanogenic [37]. In a study by [47], it was demonstrated that the bacteria carrying out the process of digestion are capable of functioning in the pH = 5.5–8.5 range, with an optimum range of 6.5–8.0. The change in pH is due to changes in VFA concentration [6]. Previous reports pointed out that the VFA could be significantly affected by the pH of the anaerobic digester: at low pH, the main VFA are acetic and butyric acids, while acetic and propionic acids played a dominant role when pH amounted to 8.0. Moreover, both the type of acid-producing bacteria and the number of bacteria could be controlled by controlling the pH [6,12,29]. Numerous studies indicate that despite the application of mesophilic conditions (35 °C), an acidic pH can occur at the hydrolysis stage, inhibiting the process of biogas production [20,48]. This problem often occurs in the digestion of food waste with a high load of organic matter formed by the long carbon chains in sugar and fat. The dynamics of the "breaking down" of these chains and the use of appropriate organic loading rate concentrations are key for the success of the process.

2.3. Volatile Fatty Acids (VFAs) Concentrations

The content of volatile fatty acids, such as acetic, valerian and propionic, are products of the acidogenesis stage in AD and significantly affect the biogas obtained, including methane. The 4 $g \cdot L^{-1}$ concentration of VFA in the bioreactor manifests itself in a decrease in biogas production [37]. The optimal concentration is assumed to be in the range of 2 $g \cdot L^{-1}$ to 3 $g \cdot L^{-1}$ [49]. Along with pH, VFAs are a common indicator of the stability of the process. Studies by Ahring et al. (1995) indicate that under thermophilic conditions in a continuous stirred tank reactor (CSTR) where manure was digested, butyrate and isobutyrate were reported as VFA compounds, the concentrations of which were used to determine the instability of the system. It is also worth noting that VFA, along with ammonia, can be used as an indicator to predict the odour of food waste due to spoilage [50]. A study by Qamaruz-Zaman and Milke (2012) showed that the most intense odour was obtained at an ammonia concentration of 360 $mg \cdot mL^{-1}$ and 940 $mg \cdot mL^{-1}$ for the isovaleric acid [51].

2.4. C/N Ratio

The processes taking place in the digester of a biogas plant can be compared to those taking place in the digestive system of ruminants. Incorrectly feeding the substrate into the chamber can give rise to a number of factors unfavourable for methanogens. Stable biogas production requires the right balance of carbon and nitrogen in the substrate. Carbon provides a source of energy for microorganisms, while nitrogen is involved in the synthesis of proteins and amino acids. A range of 15–30 or 20–30:1 is considered the optimal value, because bacteria use carbon 20–30 times faster than nitrogen [52,53]. A low C/N value results in the accumulation of ammonia in the fermenter and a change in pH, which causes methanogenic bacteria to die. A high value of the C/N parameter can indicate an imbalance of nutrients in the system and contributes to the limitation of productivity due to the rapid degradation of nitrogen [52].

The C/N value is therefore used to optimise co-digestion. In a study by Schengrong et al. (2020), the co-digestion of food waste (restaurant leftovers, which consisted of cooked rice, pork leftovers and vegetable oil), had different efficiencies due to C/N interactions vs. protein, lipid and carbohydrate compositions, indicating the need for proper selection of the reaction mixture ratio [54]. A methane quantity of 595 $mL \cdot g^{-1}$ VS (VS—volatile solids) was obtained in the study with a C/N value of 25 and a lipid/carbohydrate/protein (LCP) ratio of 63.25/22.62/14.13. Meanwhile, the methane quantity obtained with the C/N value of 30 amounted to 592 $mL \cdot g^{-1}$ VS at LCP of 48.94/39.74/11.32.

2.5. Organic Loading Rate (OLR)

The organic loading rate is defined as the amount of raw materials fed per day per volume unit of the digester. It is an important factor in determining the recovery of biogas from waste.

Volatile solids are defined as those that are biodegradable [55]. The organic loading rate depends on the type of substance being processed into biogas. The high value of the rate indicates that the reactor requires less heating. At the same time, it reduces the need for a highly efficient reactor and enriches various bacterial species [56]. However, a high OLR value can lead to an accumulation of volatile fatty acids and ethanol, resulting in an inhibition of the process. A lower than optimal OLR value has an inhibitory effect on the biogas generation process as a result of AD [57]. Li et al. (2015) carried out continuous batch experiments (40 L) to investigate the effects of OLR on the anaerobic mesophilic co-digestion of rice straw and pig manure [58]. The results showed that the maximum volumetric biogas production rate (VBPR) was improved with an increased OLR, which was in the range of 3.0–8.0 kg VS/($m^3 \cdot d$), and the maximum VBPR of 3.45 $m^3/(m^3 \cdot d)$ was obtained at an OLR of 8.0 kg VS/($m^3 \cdot d$); however, the VBPR decreased dramatically at an OLR of 12.0 kg VS/($m^3 \cdot d$) because of the severe accumulation of VFAs [58]. Meanwhile, Chandra et al. (2011) argued that a higher OLR could reduce the size of the digester and the capital cost of digestion [59]. Therefore, the value of OLR should be chosen so that the AD process takes place under optimal conditions and produces good, broadly understood economic results.

2.6. Hydraulic Retention Time (HRT)

The hydraulic retention time is a parameter that determines how long the substrate remains in the fermenter until it is replaced. It is the ratio of digester volume to the daily inflow of feedstock, the time of substrate availability for microbial growth and bioconversion into biogas. This parameter depends on the temperature, the type of bioreactor and the type of substrate used. Retention time decreases with the rise in temperature. For a bioreactor operating under mesophilic conditions, it is 14–40 days, and for a thermophilic reactor, it is 14–20 days [60].

It is possible to achieve the maximum growth rate of methanogens at startup by slowly increasing the amount of substrate fed. It is worth noting that bacteria with short activity have less time left to decompose the material. The so-called start-up phase is a critical point in the operation of a biogas plant. It is very important that the substrates fed have a constant composition—one or more substrates from the same batch. This ensures a stable growth of methanogenic bacteria. Fluctuations caused by different substrate compositions force the bacteria to constantly adapt, which leads to longer digestion times and reduced decomposition of organic matter [61]. It is also important to feed small amounts of substrate, otherwise the intermediates formed in subsequent phases will not decompose, and, as mentioned earlier, acidification of the process will occur.

3. Bioreactor Configurations

Bioreactors are devices that make it possible to obtain biogas under strictly defined process conditions, making the process independent of external conditions, which enables it to be carried out efficiently and reproducibly. Many variants and technological solutions are available on the market, allowing for the adjustment of the optimal parameters to the substrate used. Bioreactors play an important role in the growth and activity of microorganisms [60]. Due to the multiplicity of technological solutions, it is possible to divide reactors according to the number of suspended solids (total solids, TS), the mode of operation and the number of bioreactors operating simultaneously within one technological process.

3.1. Wet and Dry Anaerobic Digestion System

Depending on the TS (total solids) value, the process can be dry (TS = 20–40%) or wet (TS < 15%) [62]. The dry AD process is associated with operating at higher TS values than is the case with the wet process, which allows for more waste to be processed per bioreactor volume. However, processes carried out as dry AD have lower methane yields [63]. Moreover, it has been pointed out that the dry process is characterised by shorter retention times and lower water consumption compared to a wet system [64]. Meanwhile, analysis of the wet process emphasises the higher cost of digestate dewatering and the larger reactor capacity required for the wet process [65]. In the evaluation of the mixing conditions of the two processes, it was noted that in the dry process, it is not possible to thoroughly mix the inoculum with the substrate. This is important in terms of the availability of the cells to the medium, the efficiency of the bioconversion process and the prevention of matter putrefaction. A wet process, meanwhile, allows not only for thorough mixing but also, through the use of process water, for diluting the inhibitors [66]. Studies conducted in both dry AD and wet AD processes for sweet potato vine [67] indicate that the methane yield ($m^3 \cdot kg^{-1}$ VS_{Fed}) for the dry process was 0.25 ($m^3 \cdot kg^{-1}$ VS_{Fed}) and for the wet process was 0.32 ($m^3 \cdot kg^{-1}$ VS_{Fed}), with OLRs of, respectively, 4.6 and 0.9. Below is a diagram (Figure 1) juxtaposing a wet reactor and a dry reactor, using a single–stage reactor as an example.

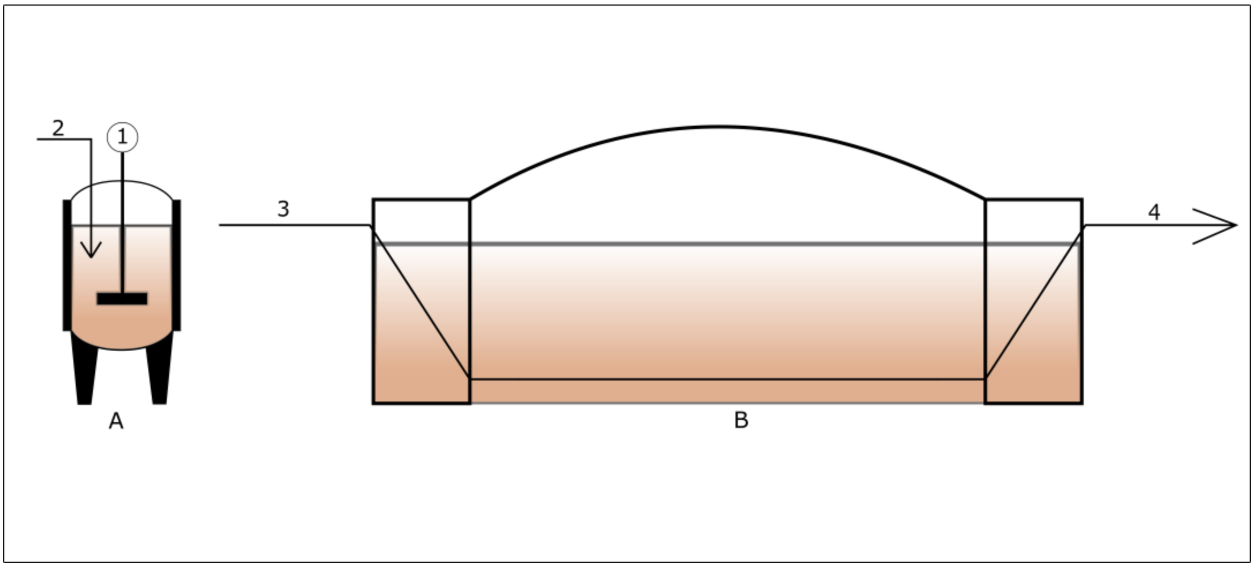

Figure 1. Wet (A) and dry (B) anaerobic reactor working in a single-stage process: 1—electric stirrer drive, 2—feed (TS > 15%), 3—feed input (TS = 20–40%), 4—feed output (scheme made by authors, partly based on [68]).

The wet reactor (A) provides a solution similar to the system that has been proven in wastewater treatment plants used for the anaerobic stabilisation of biosolids, which is an advantage, as is its mechanisation. The disadvantage, on the other hand, is the fact that the raw material fed into the reactor requires complex pretreatment. The undoubted advantage of the dry reactor (B) is the absence of rotating elements in the reactor, as well as a smaller VS loss during processing. Its disadvantage is the formation of a greater number of contaminants.

3.2. Operating Modes of an Anaerobic Digester

Anaerobic digesters can operate in different modes. The figure (Figure 2) shows a schematic drawing of a batch (C) and continuous (D) reactor:

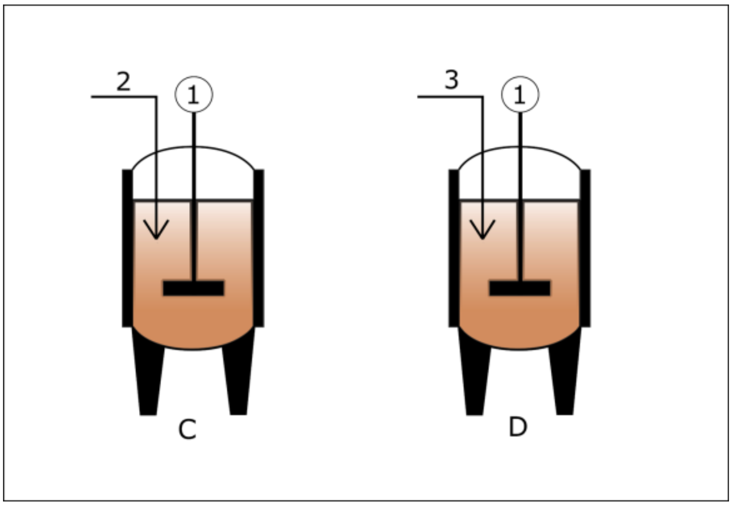

Figure 2. Batch (C) and continuous (D) working mode reactor; 1—electric stirrer drive, 2—periodically feed and 3—continuous feed (scheme made by authors, partly based on [68]).

The batch mode (see Figure 2C) is a batch system, which involves feeding the reactor with FW once and then adding inoculum to it and closing the system to produce methane. The process has low investment and maintenance costs [65]. Reactors of this type are used mainly in research work. The advantages of the batch system include low water consumption, simplicity and low purchase price. The disadvantage is the low value of the OLR parameter and the risk of reactor explosion during emptying. In turn, continuous mode (Figure 2D) is a system in which food waste is supplied to the system continuously with simultaneous removal of the digested waste before a fresh portion is added. Both biogas recirculation and the use of a continuous agitator are permitted in this method [69]. The system is designed for continuous operation. The system's advantages include low requirements for the volume of the process vessel in which biogas is collected, high quality of the biogas obtained, continuous feeding of biomass with a simultaneous generation of biogas and digestate [68], a low demand for water and less need to supply heat to the reactor [70].

3.3. Single and Multi-Stage Anaerobic Digester

Bioreactors can be combined into systems, which makes it possible to separate the stages of anaerobic digestion. The diagram below (see Figure 3) shows one-, two- and three-stage setups.

In the anaerobic digestion of food waste, we can distinguish single-stage (E) and multi-stage systems, which, in turn, can consist of two stages (F) and three stages (G). A single-stage system (E) is a system in which all stages of biogas production are carried out within a single bioreactor. For food waste, the process can take place under mesophilic conditions and thermophilic conditions. The advantages of the single-stage (E) process over multi-stage systems are the lower capital requirements and the simplicity of operating the system. Its disadvantages, on the other hand, are the higher HRT parameter value and the lower OLR value. Single-stage processes are also less efficient than multi-stage processes [60]. The two-stage system (F) performs the biogas production process in two bioreactors, where hydrolysis, acidogenesis and acetogenesis take place in the first bioreactor, while methanogenesis is carried out in the second. The advantages of the two-stage process are higher efficiency of the biogas production process, greater stability and the possibility of carrying out the process for substrates with a high OLR value. However, the two-stage process may accumulate organic acids, and an operational break is required between stages of the biogas production process. In recent years, three-stage (G) systems have been proposed, with hydrolysis of large particles occurring in the first bioreactor, acidogenesis in the second and methanogenesis in the third. As was confirmed by Salsali et al. (2005), taking the example of the digestion of activated sludge from wastewater, the three-stage process (G) achieves higher biogas yield and VS reduction compared to the two-stage setup [71]. However, such a measure generates high operating costs (related to operation and maintenance) as well as investment costs [72]. In recent years, it has been noted that a three-stage setup does not produce more methane than a two-stage setup [70].

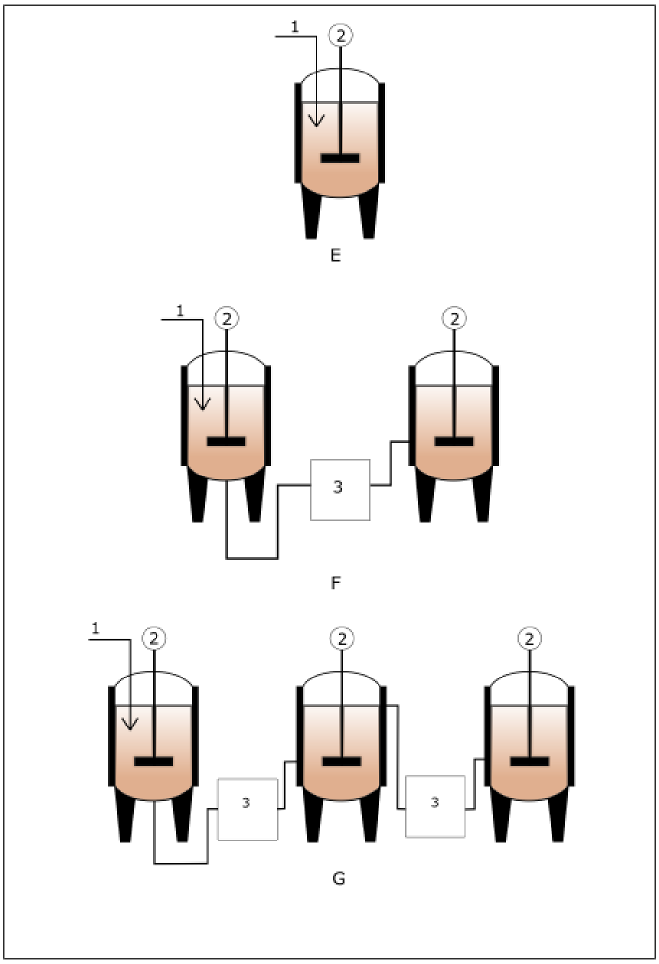

Figure 3. Single stage (E) and multi-stage (two-stage—(F), three-stage—(G)) anaerobic digestion system: 1—feed, 2—electric stirrer drive, 3—buffer tank (scheme made by authors, partly based on [68]).

4. Inhibitors

As a biotechnological process, the anaerobic digestion process is susceptible to a reduction in the efficiency of the ongoing process due to the presence of inhibitors. These are compounds, factors and conditions that cause a decline in the efficiency of the process or stop the process altogether. Any substrate component supplied in a high concentration can have an inhibitory effect on the process. Food waste is a good raw material for biogas production using anaerobic respiration. VFA and ammonia are the main factors affecting the decline in the productivity of the process. However, food waste is a mixture of different compounds that limit the process. In addition to the main factors, the influence of long-chain fatty acids (LCFAs), sulphide content, light metal ions (Mg, Ca, K, Na) and heavy metal ions (Pb, Zn, Cr, Cu, Ni) is also observed [31].

High biodegradability and rapid hydrolysis are associated with the presence of excessive amounts of VFA and ammonia, which are cited as major factors limiting efficiency [16]. In the process of dry co-digestion of food waste and pig manure, the impact of volatile fatty acids at a concentration of 16.5–18.5 $g \cdot L^{-1}$ on the decrease in the efficiency of the methane extraction process was cited as the main inhibitor [73]. It has been suggested that the low C/N ratio and high biodegradability are the causes of rapid acidification, which results in a lower methane output [74]. Propionate is cited as the main compound acidifying the AD environment.

Ammonia in anaerobic digestion is the result of protein breakdown. In low concentrations, it is essential for the functioning of anaerobic bacteria, while its high concentrations act as an inhibitor in the process. This compound exists in two forms, namely ammonia (NH_3) and ammonium ion (NH_4^+). The former is more toxic than the ammonium ion form [31]. The formation of ammonia is influenced by the temperature of the ongoing digestion—the higher the temperature, the faster the transformation of organic matter takes place, resulting in a larger amount of ammonia obtained. Anaerobic digestion of food waste has been shown to be susceptible to the inhibitory effect of ammonium ion at 2 $g \cdot L^{-1}$ [75]. It has also been pointed out that NH_3–N content ranging from 1.7 $g \cdot L^{-1}$ to 14 $g \cdot L^{-1}$ can reduce methane yield by 50% [31].

The content of LCFAs has an inhibitory effect on the process of anaerobic respiration, and their presence in the fermented substrate is due to the presence of fats. It has been pointed out that the presence of oleic acid and palmitic acid at concentrations of 4.5 $g \cdot L^{-1}$ and 3.0 $g \cdot L^{-1}$, respectively, caused a decrease in biogas output >50% [76]. The inhibitory effect of LCFAs is influenced by the temperature of the process. Under mesophilic conditions, their formation is more limited than under thermophilic conditions due to the lower temperature of the ongoing process.

The presence of sulphide (H_2S) in food waste has a negative impact on the process of methane formation, especially when conducted under thermophilic conditions. The sources of sulphides include, for example, slaughterhouse waste and food waste [31]. It has been pointed out that protein is the main source of H_2S in food waste undergoing the digestion process [77]. The sulphide inhibition effect is caused by the competition for the carbon and hydrogen available in the system between sulphate-reducing bacteria (sulphate-reducing bacteria, SRB) and methane–forming bacteria during the AD process. Sulphur compounds affecting the inhibition of methanogenesis are H_2S (hydrogen sulfide), S_2^- (sulfide anion) and Na_2S (sodium sulfide) at concentrations of, respectively, 50 $mg \cdot mL^{-1}$, 100 $mg \cdot mL^{-1}$ and 160 $mg \cdot mL^{-1}$. In the case of H_2S, the concentration value causing inhibition can be between 600 $mg \cdot mL^{-1}$ and 1000 $mg \cdot mL^{-1}$, which is due to the occurrence of bacteria characterised by resistance to high concentrations of H_2S [31,75].

Light and heavy metal ions can have an inhibitory effect on biogas production. Ca^{2+}, Mg^{2+}, K^+ and Na^+ ions are supplied to the bioreactor in the form of pH-regulating compounds or are the products of decomposition of organic matter. Low concentrations are beneficial to the process. It has been pointed out that the limiting concentrations of light metal ions are, respectively, 2.8 $g \cdot L^{-1}$ for Ca^{2+}, 2.4 $g \cdot L^{-1}$ for Mg^{2+}, 3 $g \cdot L^{-1}$ for K^+ and 6–30 $g \cdot L^{-1}$ for Na^+ [31]. Heavy metal ions, which include lead (Pb), zinc (Zn), chromium (Cr), copper (Cu) and nickel (Ni), show inhibitory effects at concentrations of, respectively, 340 $mg \cdot L^{-1}$ for Pb, 400 $mg \cdot L^{-1}$ for Zn, 130 $mg \cdot L^{-1}$ for Cr, 40 $mg \cdot L^{-1}$ for Cu and 10 $mg \cdot L^{-1}$ for Ni. Heavy metal ions show the ability to inhibit anaerobic digestion at lower concentrations than light metal ions. The mechanism of inhibiting the process by heavy metal ions involves disruption of the structure and inhibition of enzyme function; however, in the case of AD of food waste, heavy metals are below threshold values. Light metal ions (Na^+, K^+), the concentrations of which can contribute to the inhibition of the fermentation process, may be the problem [78].

5. Pretreatments

In order to increase the efficiency of biogas production, including methane, various measures are being taken to improve the contact between the microflora and the biomass that is being digested. The efficiency of AD of food waste is significantly influenced by the conversion of biomass or, more specifically, the efficiency of the transformation of compounds at each stage of the process and also the availability of the medium [53,79]. Hydrolysis is the rate-limiting stage in anaerobic biodegradation as a whole, as it is the most problematic stage, especially when vegetable substrates, but also food waste of various origins, are used. At this stage, polymers are converted into soluble monomers by enzymatic hydrolysis. The reaction is catalysed by extracellular enzymes of microorganisms called hydrolases or lyases [80]. However, not all substances, especially those that belong to the so-called lignocellulosic biomass, yield to hydrolytic enzymes with equal ease. Hence, there is a need to take measures to support the hydrolysis stage. It is assumed that appropriate technological pretreatment of the feedstock can significantly increase the efficiency of anaerobic digestion, with little additional energy and cost input associated with its initial processing.

There are many factors, for example, particle size, material structure and substrate composition, that can affect the rate of hydrolysis, especially of high-molecular-weight compounds and granular substrates. To accelerate the first stage of biodegradation, the following methods can be used: the common grinding method, i.e., ultrasonic, mechanical grinding, thermal, chemical, microwave or a combination thereof, and biological pretreatment [81–88]. Food waste consists mainly of carbohydrate polymers (starch, cellulose, hemicellulose), lignin, other organic substances (proteins, lipids, acids, etc.) and the remaining, smaller inorganic part. Most carbohydrate and protein polymers are found in solid form, such as rice, vegetables and meat. Mechanical pretreatment in their case has a significant impact on improving AD performance. Scientists testing this kind of pretreatment report that [53] AD batch experiments revealed that when particle size was reduced by bead milling at 1000 rpm, the methane yield increased by 28%. There was a reduction in FW particle diameters from 0.843 to 0.391 mm due to pretreatment with a ball mill and an increase in solubility by 40% of the total COD. Meanwhile, Izumi et al. additionally proved that excessive reduction of FW particle size may not promote methane production, as dynamically degrading substrates with much smaller particles contributes to the accumulation of VFA.

A lot of attention in the literature has been given to chemical pretreatment, carried out during the hydrolysis stage. Chemical treatment of cellulosic material and starch components can release sugar and increase its availability to bacteria by breaking the glycosidic bonds of polysaccharides [88]. Among chemical pretreatments, alkaline and acid pretreatment, using alkalis and acids, respectively, are distinguished. Alkaline pretreatment uses bases such as sodium hydroxide, calcium hydroxide, potassium hydroxide and ammonia water, while acid pretreatment uses organic and inorganic acids such as sulphuric (VI), hydrochloric, nitric (V), phosphoric (V), acetic and maleic acids. Due to environmental reasons, acids are more often used in high dilutions but under elevated temperature conditions. For the same reasons, chemical compounds used for processing must be recovered once the process has been completed. Vavouraki et al. (2013) report that pretreatment with 1.12% HCl for 94 min or 1.17% HCl for 86 min (at 100 °C) can increase the concentration of soluble sugars by 120%, compared to untreated substrate [88]. Similar effects have been demonstrated by previously reported chemical pretreatment in combination with ultrasound and steam, leading to the breakdown of glycosidic bonds in starch [89,90]. The implementation of the pretreatment techniques discussed above has its drawbacks, despite the increase in biogas/methane generation efficiency. For example, during chemical pretreatment, carboxylic acids, furans and phenolic compounds were released, resulting in AD inhibition and reduced biogas production [91,92]. Alkaline pretreatment is one of the best practices for dissolving complex matter, with the following order of effectiveness: NaOH > KOH > $Mg(OH)_2$ and $Ca(OH)_2$ [80]. However, too high a concentration of Na^+ or K^+ can cause subsequent AD inhibition. Alkaline pretreatment is also often conducted in combination with thermal or ultrasonic pretreatments, yielding more favorable results than the application of one type of method. It should be noted that pretreatment usually results in higher capital costs due to additional energy and additive (chemical) inputs [93]. In practice, the amount of methane produced when this type of pretreatment is involved is often insufficient to compensate for the additional cost, making its application uneconomical [89].

As a rule, the biological pretreatment method uses selected species of fungi (*Ceriporiopsis subvermispora, Trichoderma reesei*), yeast, bacteria (*Clostridium thermocellum*) and enzymes [94]. Compared to physical and chemical pretreatment methods, energy inputs in biological pretreatment are significantly lower. Fungal processing is carried out under appropriate conditions: temperature is in the range of 27–37 °C, while incubation time amounts to 8 days to 12 weeks [53,82]. However, the relatively long biological pretreatment time may limit the application of this method on a technical scale. A problematic competition for carbohydrates between microbes involved in pretreatment and anaerobic bacteria may also occur. Pretreatment with enzymes is a method that is researched and used in practice more frequently [82]. For example, Moon and Song (2011) tested the effect of enzymatic solubilisation of FW in methane production in an upflow anaerobic sludge blanket reactor (UASB reactor) using an optimised mixture of carbohydrase: protease: lipase (1:2:1) enzymes, respectively, 0.2% (*w*/*w* FW) of mixture dose and 10 h hydrolysis reactions [95]. The results were up to 95% of soluble COD (chemical oxygen demand) removal efficiency with an observed methane gas yield of 350 mL $CH_4 \cdot g^{-1}$ soluble COD and 9.1 g soluble COD/L/d of OLR. However, it should be stressed again that the biogas/methane yields of most biogas plants will not offset the cost of purchasing enzymes, making it unjustifiable to use them on a technical scale.

6. Anaerobic Co-Digestion

Due to its high organic content (lipids, proteins, carbohydrates, starch, cellulose) as well as high water content and biodegradability, food waste (FW) has been identified as a promising substrate for biogas and methane production through anaerobic digestion (AD) [12,96]. Nevertheless, it has been shown that in many cases, the mono-digestion of FW has resulted in inhibition of the process and minimal production of the mentioned gases. The main problem in using FW as the sole substrate in digestion is the inadequate C/N (carbon to nitrogen) ratio [97]. Rodríguez-Jiménez et al. (2022) [98] report that a too high C/N ratio leads to a shortage of nitrogen, which is an essential element for microbial biomass biosynthesis. On the other hand, a too high C/N ratio contributes to the intense proliferation and activity of microorganisms, resulting in excessive production of ammonia and pH change in the bioreactor [16]. Similarly, very high concentrations of lipids in food waste result in the accumulation of excessive amounts of LCFA and VFA, which in turn acidify the fermenter environment [53,96,98]. The lack of essential trace elements (e.g., nickel, cobalt, molybdenum, iron, selenium and tungsten for methanogenic bacteria and zinc, copper and manganese for hydrolytic bacteria) in the starting substrate reduces the number of microorganisms responsible for the digestion of waste to generate biogas, including methane [10,60,99]. Moreover, the decomposition of FW with too high protein concentrations leads to the formation of excessive foam in the bioreactor, which also inhibits the production of the gases in question [10,53].

In order to minimise the risk of the problems described above, food waste is increasingly being co-digested together with a co-substrate. The optimal substrate/co-substrate mass ratio in the mixture and the interaction of these two elements enable the optimisation of the entire digestion process and the production of significantly higher amounts of biogas (including methane) than in the case of separate digestion of individual components (see Figure 4) [12,16,60,97,98,100]. The results of a number of studies conducted around the world have shown that promising co-substrates for FW are plant biomass, livestock faeces and wastewater and sewage sludge.

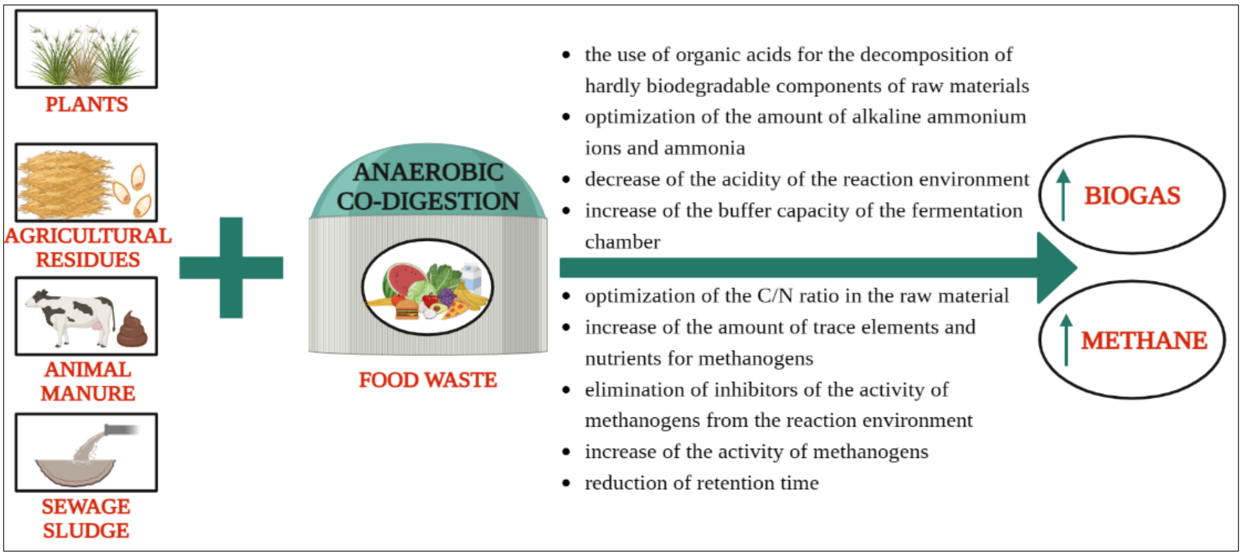

Figure 4. Effects of anaerobic co-digestion of food waste with different substrates (author's own scheme).

6.1. Co-Digestion of Food Waste with Plants, Plant Residues and Algae

The above observations are confirmed by a study of Chen et al. (2016), which showed that co-digestion of food waste (FW) and tall fescue (Tf) increased biogas yields, including methane, and substrate degradation efficiency compared to the mono-digestion of FW and Tf [101]. According to the authors, the increased production of the gases in question was due to a synergistic effect between the substrates. This is because the addition of plants increased the C/N ratio of the raw material to the optimal value for anaerobic digestion. In turn, the large number of organic acids from the decomposition of food waste induced chemical treatment of lignocellulosic biomass, accelerating the degradation of cellulose, hemicellulose and lignin of tall fescue. In addition, it was pointed out that the use of acids for the decomposition of plants reduced the accumulation of these compounds in the bioreactor and minimised the occurrence of the problem of over-acidification of the ferment, which in many cases ends up inhibiting the process. Similarly, Oduor et al. (2022) proved that co-digestion of food waste and water hyacinth (WH) increased the C/N ratio of the feedstock and the buffer capacity of the digester [102]. These changes led to a decrease in the acidity of the environment, leading to an increase in the activity of methanogenic bacteria, which in turn allowed biogas production to be higher by 258.16 mL$\cdot$g^{-1} VS and by 359.74 mL$\cdot$g^{-1} VS, compared to the mono-digestion of FW and WH.

Ferreira et al. (2021) showed that a mixture of food waste and microalgae (MA) increased methane yields, compared to the separate digestion of FW and MA, by 21% and 55%, respectively [103]. Based on the results, it was concluded that the increase in the production of the gas in question was due to an increase in the buffer capacity of the ferment in the digester and a decrease in the accumulation of organic acids through the addition of microalgae. Similar results were presented by Zhao and Ruan (2013), who found that co-digestion of food waste and algae from Lake Taihu resulted in an increase in protease and dehydrogenase activities of methanogenic microorganisms and an increase in biogas yield by 60.1 mL$\cdot$g^{-1} TS and 86.5 mL$\cdot$g^{-1} TS, respectively, compared to the mono-digestion of FW and algae [104]. According to the authors, these changes were caused by the increase in the buffer capacity of the digester and the selection of the optimal C/N value of the substrates. The right ratio of carbon to nitrogen led to the reduction in volatile fatty acids and the maintenance of suitable conditions for the growth of methanogenic bacteria, which led to an increase in their enzymatic activity and faster conversion of substrates into biogas.

At the same time, Zhang et al. (2018) report that the use of food waste as a co-substrate in anaerobic digestion of tree leaves and twigs (yard waste—YW) accelerated the degradation of cellulose and hemicellulose by 6.5–20.3% and 14.8–53.1%, respectively, compared to the control [105]. Increasing the extent of polysaccharide degradation allowed methane yields to increase by 47% and 21%, respectively, compared to the mono-digestion of YW and FW. Moreover, the study by Chen et al. (2014) showed that the addition of grass cuttings and leaves to the digestion mixture of food waste led to a reduction in process retention time, compared to the separate digestion of each substrate [106]. In addition, it was found that increasing the share of food waste in digested substrates led to an increase in biogas yield, while increasing the share of green waste in the feedstock led to an increase in biogas production efficiency.

6.2. Co-Digestion of Food Waste with Crop Residues

Yong et al. (2015) report that co-digestion of food waste and a mixture of maize, sorghum and wheat straw increased methane production efficiency by 39.5% and 149.7%, respectively, compared to the mono-digestion of FW and straw [107]. According to the above authors, the addition of straw balanced the C/N ratio in the ferment, which minimised the nitrogen content in the bioreactor to the optimal value for the proper course of the process. In addition, it was found that the use of crop residues as a co-substrate reduced the concentration of cations in the digester (e.g., sodium ions), which are potential inhibitors of the growth of methanogenic microorganisms. The study by Pei et al. (2014) showed that a mixture of food waste and rice straw (RS) increased biogas and methane production and reduced process retention time [108]. Based on the analytical results obtained, it was concluded that the increase in the amount of the gases in question was due to the maintenance of the proper level of volatile fatty acids and the reduction of ammonia concentration in the bioreactor. For this study, the methane yields of co-digestion were 178% and 70% higher than those of mono-digestion of FW and RS.

Owamah and Izinyon (2015), on the other hand, proved that a mixture of food waste and maize husks (MH) increased biogas production by 17.82 $L \cdot g^{-1}$ VS and 23.68 $L \cdot g^{-1}$ VS compared to the mono-digestion of FW and MH [109]. According to the above authors, the addition of crop residues as a co-substrate increased the C/N ratio of the feedstock, which made it possible to minimise the occurrence of the problem of over-acidification of digested substrates at the initial stages of the process. In addition, it was indicated that the selection of a high OLR contributed to an increase in the number of methanogenic bacteria in the digester, which in turn enabled faster and more efficient degradation of the substrate and the production of higher volumes of the gases in question.

In addition, Haider et al. (2015) reported that co-digestion of food waste and rice husks (RH) increased biogas yields compared to the mono-digestion of FW and RH [110]. Based on the results, it was concluded that the increase in the production of the gas in question was due to the maintenance of the proper pH of the environment and the correct level of volatile fatty acids in the bioreactor, as well as increasing the C/N ratio of the feedstock to the optimal value for anaerobic digestion.

6.3. Co-Digestion of Food Waste with Animal Faeces, Wastewater and Sewage Sludge

The study by Rahman et al. (2021) showed that the addition of chicken manure to food waste digested at 28 °C and 37 °C increased the rate of cumulative biogas production by 84 mL and 388 mL, respectively, compared to the mono-digestion of FW [111]. Based on the analytical results obtained, the authors concluded that the increase in the production of the above gases was due to the synergistic effect between the feedstocks. Food waste is an easily biodegradable substrate whose rapid hydrolysis very often leads to strong acidification of the reaction environment, which in turn inhibits the growth of methanogenic microorganisms. Meanwhile, manure is very rich in biodegradable proteins and urea, whose degradation generates alkaline ammonium ions and ammonia, thus stabilising the pH of the ferment and increasing its buffer capacity. According to the authors, the relationship discussed above and the correct C/N ratio led to an optimal reduction in the acidity of the environment, thus leading to an increase in methanogen activity and consequently optimising biogas and methane production. Similar findings were presented by Chuenchart et al. (2020), who proved that co-digestion of food waste and chicken manure resulted in an increase in specific methane yield [112]. According to these authors, co-digestion of the above-mentioned organic waste buffers the reaction environment and increases the availability of nutrients (including nitrogen), which in turn allows a synergistic effect to occur between microorganisms that convert organic waste sequentially to long-chain fatty acids, acetates, carbon dioxide and hydrogen and microorganisms that convert the resulting CO_2 and H_2 to methane through proton reduction.

Zhang et al. (2013) also reported that co-digestion of food waste and manure, in this case cattle manure (CM) increased biogas and methane yields by 551 $mL \cdot g^{-1}$ VS and 405 $mL \cdot g^{-1}$ VS, respectively, compared to the mono-digestion of CM [113]. Based on the analytical results obtained, it was shown that the addition of manure as a co-substrate balanced the C/N ratio in the raw material, which made it possible to use high OLR values. The ammonia produced by manure decomposition increased the biodegradation of long-chain fatty acids and lipids, thus neutralising the reaction of the environment. In addition, it was indicated that cattle faeces provided trace elements such as manganese, zinc, magnesium and calcium, which are essential for the initiation of methanogenic enzyme activity. Similar research conclusions were presented by Agyeman and Tao (2014), proving that a mixture of food waste and cow manure resulted in an increase in methane production efficiency and yield. In addition, the authors observed that the dewatering capacity of the digestate increases as the particle size of the waste used as feedstock decreases [114].

A study conducted by Cour Jansen et al. (2004) found that the addition of wastewater along with sewage sludge to food waste increased methane production efficiency by 45–75 N m^3/Mg^{-1} VS compared to the mono-digestion of sludge [115]. Similar conclusions were drawn by Kim et al. (2003), who analysed the efficiency of methane production during the digestion of sewage sludge (SS) and food waste [116]. The above-mentioned authors indicated that co-digestion increased the production of the gas in question by 99–117 $L \cdot g^{-1}$ VS compared to the mono-digestion of SS. In contrast, Heo et al. (2004) report that the addition of waste-activated sludge to food waste not only increased the buffer capacity of the digester but also balanced the amount of nutrients present in the reaction environment [117]. In addition, it was shown that as the amount of FW in the mixture increased, the methane yield also increased.

In addition, Pilarska et al. in their publications emphasizes the stabilizing effect of digested sewage sludge (as inoculum) on the AD process [13,20,26]. The beneficial effect of sewage sludge is due to its high buffering capacity related to alkalinity, generated by the carbonates and bicarbonates present in the sludge. The mechanism of formation of these compounds as a result of nitrogen transformations taking place during sludge fermentation was described by the scientist [1]. The same author also discovered a synergy between confectionery waste (wafer waste) and raw sewage sludge. The AcoD of both of these substrates results in biogas with a high content of methane [118]. This phenomenon was attributed to a significant number of bacterial cells in the sewage sludge (where their multiplication was intensified during the process) and high enzymatic activity of the fermented confectionery waste.

7. Efficiency of Anaerobic Digestion of Food Waste

The efficiency of anaerobic digestion and co-digestion of food waste depends, among other things, on the composition and properties of the substrates used. Food waste generated at different locations and at different stages of the food chain is characterised by its unique, highly complex and heterogeneous composition [100]. Xu et al. (2018) [99] and Negri et al. (2020) [119] report that the characteristics and composition of FW belonging to the same category (e.g., restaurant waste, household kitchen waste, fruit and vegetable residues) are highly variable mainly due to the diet and eating habits of the people who produce the waste in question. In addition, the above-mentioned authors point out that the composition of FW is also influenced by culture, geographic origin, climate, season, source and treatment method of waste and availability of food products.

According to Xu et al. (2018), the highest methane production potential is found in lipid-rich waste (methane potential: 1.014 $m^3 \cdot kg^{-1}$ VS), followed by proteins (methane potential: 0.74 $m^3 \cdot kg^{-1}$ VS) and carbohydrates (methane potential: 0.37 $m^3 \cdot kg^{-1}$ VS) [99]. On the other hand, the lowest potential is shown by lipid-poor substrates that have a high content of hard-to-biodegrade lignocellulosic biomass (methane potential: 0.16–0.35 $m^3 \cdot kg^{-1}$ VS). The content of individual organic components in food waste should provide optimal conditions for the growth of methanogenic microorganisms, i.e., an optimal balance between the available nutrients (optimal carbon to nitrogen ratio—C/N) [100]. According to Zhang et al. (2014) [12], the optimal C/N value for proper and rapid anaerobic digestion of FW is in the range of 20–30, while according to Uddin et al. (2021), the value should be between 25 and 35 [120]. A study conducted by Ferreira et al. (2021) shows that the highest methane yield (241 $mL \cdot g^{-1}$ VS) was obtained for the digestion of lipid-rich waste (rice, meat, beans), for which the C/N ratio was 11.6 [103]. In contrast, the lowest methane production (35 $mL \cdot g^{-1}$ VS) was demonstrated by digestion of substrates rich in carbohydrates and hard-to-biodegrade components (lettuce, carrots, tomato), for which the C/N ratio was 9.4. Similar results were obtained by Li et al. (2013), who indicated that digestion of fruit and vegetable residues gave a methane yield of 342 $mL \cdot g^{-1}$ VS (C/N ratio: 13.1), whereas digestion of kitchen waste, characterised by a more complex and diverse composition, gave a methane yield of 541 $mL \cdot g^{-1}$ VS (C/N ratio: 20.3) [121]. Therefore, it seems virtually impossible to define a universal and uniform range of methane potential for the highly complex and diverse group of substrates that make up food waste. In order to demonstrate the differences in the efficiency of food waste digestion, Tables 1 and 2 present information on the yield of biogas and methane from different waste groups, taking into account the different values of the temperature and C/N ratio (see Table 1) and different waste groups (see Table 2).

Table 1. Efficiency of anaerobic digestion of food waste: the results for different values of temperature and C/N ratio.

Substrates	Conditions	Results	References
kitchen waste	• temperature: 55 ± 1 °C • C/N ratio: 24.08	specific biogas yield: 12.6 mL/gTSd	[104]
kitchen waste	• temperature: 37 °C • C/N ratio: 20.3	methane yield: 541 mL/gVS	[121]
student dormitories kitchen waste	• temperature: 28 °C • C/N ratio: 35.2	the rate of cumulative biogas: 312 ± 9 mL	[111]
student dormitories kitchen waste	• temperature: 37 °C • C/N ratio: 35.2	the rate of cumulative biogas: 532 ± 17 mL	[111]
canteen food waste	• temperature: 37 °C • C/N ratio: 17.28	biogas yield: 357.85 mL/gVS	[102]
canteen food waste	• temperature: 37 ± 1 °C • C/N ratio: 14.4	methane yield: 326.4 mL/gVS	[106]
canteen food waste	• temperature: 37 ± 1 °C • C/N ratio: 13	biogas yield: 11.10 L/gVS	[109]
canteen food waste	• temperature: 35 ± 1 °C • C/N ratio: 21.1	biogas production: 621 mL/gVS methane yield: 410 mL/gVS	[113]

Table 1. *Cont.*

Substrates	Conditions	Results	References
restaurant food waste	• temperature: 55 °C • C/N ratio: 21.2	average daily biogas: 224 L/d methane production: 120 L/d	[112]
restaurant food waste	• temperature: 37± 1 °C • C/N ratio: 28.2	methane yield: 573 mL/gVS	[122]
leftovers of cooked foods, meats, rice, breads, noodles and vegetables	• temperature: 35 °C • C/N ratio: 28.4	biogas production: 0.49 m^3/kgVS methane production yield: 0.281 m^3/kgVS	[107]
vegetables, fruits, rice, noodles, meat, fish and eggs	• temperature: 35 ± 1 °C • C/N ratio: 17.28	biogas yield: 674.40 NmL/gVS	[123]
rice, beans and meat	• temperature: 35 ± 1 °C • C/N ratio: 11.6	methane yield: 241 mL/gVS	[103]
lettuce, carrots, and tomato	• temperature: 35 ± 1 °C • C/N ratio: 9.4	methane yield: 35 mL/gVS	[103]
orange peel, banana peel and papaya peel	• temperature: 35 ± 1 °C • C/N ratio: 24.2	methane yield: 44.8 mL/gVS	[103]
fruit and vegetable waste	• temperature: 37 °C • C/N ratio: 13.1	methane yield: 342 mL/gVS	[121]

Table 2. Efficiency of anaerobic co-digestion of food waste with other substrates.

Co-Substrates	Conditions	Results	References
tall fescue	• temperature: 37 ± 1 °C • FW/Tf ratio: 1.52	biogas yield: 406 mL/gVS methane yield: 296.01 mL/gVS	[101]
water hyacinth	• temperature: 35 °C • mixture ratio: 30 (FW):70	biogas yield: 616.01 mL/gVS	[102]
microalgal biomass	• temperature: 37 °C • mixture ratio: 75 (FW):25	methane yield: 514 mL/gVS	[103]
Taihu algae	• temperature: 55 ± 1 °C • C/N ratio: 15:1	specific biogas yield: 14.9 mL/gTSd	[104]
yard waste	• temperature: 37 °C • mixture ratio: 9:1 (FW)	cumulative methane yield: 131 mL/gVS	[105]
green waste	• temperature: 37 °C • mixture ratio: 60 (FW):40	biogas yield: 390.2 mL/gVS methane yield: 272.1 mL/gVS	[106]
straw of maize, sorgos and wheat	• temperature: 35 °C • mixture ratio: 5 (FW):1	biogas production: 0.58 m^3/kgVS methane production yield: 0.392 m^3/kgVS	[107]
rice straw	• temperature: 35 °C • mixture ratio: 3 (FW):1	methane yield: 60.55 mL/gVSd	[108]

Table 2. *Cont.*

Co-Substrates	Conditions	Results	References
maize husk	• temperature: 37 °C • mixture ratio: 75 (FW):25	biogas yield: 28.92 L/gVS	[19]
rice husk	• temperature: 37 °C • mixture ratio: 10.5 (FW):1	specific biogas yield: 584 L/kgVS	[20]
poultry manure	• temperature: 37 °C • mixture ratio: 2 (FW):1	the rate of cumulative biogas: 920 ± 11 mL	[109]
cattle manure	• temperature: 35 °C • FW/CM ratio: 2	biogas production: 570 mL/gVS methane yield: 388 mL/gVS	[113]
pig manure	• temperature: 37 °C • mixture ratio: 4 (FW):1	specific methane yield: 521 mL/gVS	[124]
sewage sludge	• temperature: 34 °C • mixture ratio: 20 (FW):80	methane yield: 326 $Nm^3/tonVS_{in}$	[115]
sewage sludge	• temperature: 3 °C • mixture ratio: 50:50	methane yield: 0.215 L/gVS	[116]
sewage sludge	• temperature: 55 °C • mixture ratio: 50:50	methane yield: 0.280 L/gVS	[116]
waste activated sludge	• temperature: 35 °C • mixture ratio: 90 (FW):10	specific methane production: 0.346 $m^3/kgVS$	[117]

8. Development Perspectives and Recommendations

Producing biogas from food waste can significantly increase energy efficiency by recovering energy from materials that would otherwise end up in landfills or undergo other disposal processes, such as incineration. Anaerobic digestion provides the opportunity to convert FW into a valuable energy product while reducing the negative impact on the environment, in line with the principles of sustainable waste management [6,26,125].

The use of waste biomass is the most economical solution for the operation of biogas plants. Acquiring free substrates or building biogas plants at production facilities is the most promising solution today, which is being urged on in countries where the biogas market is stagnating. It is worth pointing out that agricultural production of biomass for energy purposes (for example, maize) competes with food production due to the reduction in acreage cultivated for food and livestock feed [27,46]. The problem of such competition does not exist in the case of biogas production from agricultural by-products, including, specifically, food waste. It also does not contribute to a decrease in food production due to reduced soil fertility caused by a reduction in the amount of organic matter returned to the soil when plant by-products (e.g., straw) are burnt and the consequent reduction in crop yields. The construction of new biogas plants operating "at the source" also offers broadly defined potential for the development of local communities [18,126]. It means, on the one hand, new jobs, development of the local economy and energy independence and on the other hand, building environmental awareness, integration and active participation in the transformation of the energy sector.

Recommendations for developing and improving the efficiency of the AD process of food waste involve the following: (i) maximizing recovery of food waste for biogas production; (ii) investment in infrastructure: construction of new or upgrading of existing plants; (iii) technological research and development: design of new and more efficient reactors, heat recovery methods and optimisation of the conditions for the microorganisms responsible for the process; and (iv) cooperation between the public and private sectors. These partnerships may include participation in financing, joint research, business support and infrastructure provision. Creating a favourable regulatory environment and stimulating private investment is the basis for accelerating the development of the economic sector under consideration.

9. Conclusions

Anaerobic digestion of food waste, which is the product of food processing and consumption is, due to the high organic matter and moisture content of this waste, a very effective approach. Using food waste in the AD process to produce biogas has great environmental, economic and social potential. FW can be effectively digested under both mesophilic and thermophilic conditions, although mesophilic conditions are more common on a technical scale. On the one hand, they limit the release of inhibitors, while on the other hand, they create limitations for the decomposition of high-fat substrates that require elevated temperatures. In the case of FW management, pretreatment is often necessary in order to release components that are difficult to digest for hydrolytic enzymes (such as cellulose). The use of pretreatment is sometimes not cost-effective due to the high consumption of heat and energy and the cost of purchasing chemicals and enzymes. The most popular pretreatment of FM is the mechanical, or biological, method. Anaerobic co-digestion of FW with other substrates contributes to increased biodegradation of LCFA, as well as methane yield. In addition, AcoD can also improve buffer capacity and lead to increased acceptable organic loads compared to single digestion. Progress needs to be made on new biofermenter designs, whose performance has a significant impact on the accessibility of the cells to the medium and the efficiency of the bioconversion process. Multi-stage systems, especially two-stage systems, have a great advantage over single-stage systems, as they provide greater efficiency and stability and offer the possibility of carrying out the process for substrates with high OLR values. Nevertheless, they require technical and structural refinement. The technology of managing food waste for biogas production is undoubtedly a promising method but one that requires new, effective technologies and favourable investment conditions.

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