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Review

Analysis of biological treatment technologies, their present infrastructures and suitability for biodegradable food packaging - A review

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ABSTRACT

Recently, there has been an increased demand for biodegradable plastics in the food packaging industry, especially for highly food soiled packaging items containing food/beverage solids that will not be recycled using a non-biological process. However, the increased usage of those materials have also raised concerns and confusion, as a major part of these biodegradable plastics are not effectively separated nor recycled. The lack of acceptance in recycling facilities, related to confusion with their conventional polymers counterparts, as well as short retention times of recycling facilities, often incompatible with the degradation kinetics of biodegradable plastics, stand as the major drawbacks for bioplastics treatment. Additionally, the presence of incompletely biodegraded bioplastics during biological treatments or in the final products i.e. compost or digestate, could lead to process failure or limit the commercialization of the compost. This work critically reviews the fundamentals of the biological treatments, anaerobic digestion and composting processes, and discusses the current strategies to improve their performance. In addition, this work summarizes the state-of-the-art knowledge and the impact of bioplastics on full-scale treatment plants. Finally, an overview of the current installed treatment capacity is given to show the areas of opportunity that can be improved and exploited to achieve a better waste management of biodegradable plastics.

1. Introduction

Plastics have undoubtedly revolutionized human civilization with their unique properties, including affordability, durability, and versatility. Their ubiquity in almost every aspect of modern life highlights their indispensable role in meeting various human needs. Currently, over 400 million tons of plastics are produced annually, and unfortunately, only 10% of the global plastic production is recycled (European Bioplastics, 2023; European Commission, 2022; Ghasemlou et al., 2024). In this context, international strategies, such as the Circular Economy Plan, the European Green Deal, and the European Union Plastics Strategy, have been implemented to reduce, reuse and recycle fossil-based plastics and implement the use of their biodegradable and bio-based counterparts (European Commission, 2022). In this sense, bioplastics were introduced as an alternative to fossil-based plastics with the aim to ease their degradation. However, the differences in chemical composition between bioplastics and conventional plastics limit their recycling in conventional infrastructures and their management is complicated. Nevertheless, in 2023 bioplastics production accounted for 0.5% of the total plastic production (2.18 million tons*)* (European Bioplastics, 2023).

Bioplastics are defined as a plastic that is biobased and/or biodegradable (Dilkes-Hoffman et al., 2019), and are used for a wide variety of applications (Fig. 1) from which packaging represents the largest market, with a share of over 40% of the total bioplastics market in 2023. Thereby, a holistic approach that considers not only the production and composition of bioplastics, but also their final fate, is necessary to fulfil the recycling of bioplastics under a sustainable approach (García-Depraect et al., 2021).

Bioplastics end-of-life treatment can be classified in two main methods: biological and physicochemical (Rasheed et al., 2024). The physicochemical pathway includes mechanical, chemical, and physical techniques including pyrolysis and solvolysis processing which breakdown the polymers to remake bioplastics again. Incineration also enters

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this category, however, it is the less preferable method due to its high carbon footprint. On the other hand, biological methods involve the degradation of bioplastics into monomers, which are used as a source of carbon and energy by bacteria. In this holistic sense, the polymer molecules are returned to the environment without generating toxic subproducts. However, the kinetics of biodegradation and final biodegradability of bioplastics depends on the polymer physical characteristics, polymer chemical composition, microbial community structure, etc. (Islam et al., 2024). Even though bioplastics degrade in much shorter time compared to conventional plastics, the inappropriate biodegradation conditions might result in incomplete degradation and consequently, in the production of microplastics. Thus, proper management techniques to support an effective bioplastics degradation are necessary to reduce the risk of microplastics production. For instance, compostable bioplastics are manufactured to be degraded under controlled standardized temperature and moist conditions, which can only be achieved in industrial composting. On the other hand, in home composting the conditions vary and the temperature is lower. Hence, home-composting is not ideal for bioplastics management (Ahsan et al., 2023). One factor to be considered when choosing the proper recycling technique is the quality and grade of the bioplastic waste. Typically, physicochemical techniques are used or preferred for high grade plastic or for a mix of several bioplastics types that are preferably not contaminated with other waste, i.e. food waste. Moreover, biological treatments are suitable for bioplastics contaminated with food waste pollutants i.e. odor or fat components, which are difficult to remove. Additionally, other applications where bioplastics are intended to be used and disposed together with the product i.e. tea containers, coffee bags, coffee capsules, or other beverages shall be considered for biological treatments. (Fredi and Dorigato, 2021; Jung et al., 2023). It is important to highlight that when biological treatments are suitable, it is recommended to adopt industrial processes to ensure a cost-effective bioplastics degradation.

Biological treatment includes anaerobic digestion and composting processes, which allow the recovery of bioplastics in the form of valuable products such as energy and fertilizer, respectively. However, the confusion surrounding terms like “bio-based” and “biodegradable”, along with the unproper separation from the organic fraction of municipal waste, have created unacceptance of bioplastics in recycling facilities. In this sense, test standards have been established by international agencies such as ASTM, CEN, JIS and ISO to facilitate the quantification of the biodegradability of bioplastic materials. However, these standards are not always suitable for describing the biodegradation process or predicting the biodegradability of bioplastics under real conditions. This is due to the complexity of the surrounding biotic and abiotic conditions and the variety of commercial bioplastic compositions, resulting in different degradation rates and ultimately low and unpredictable levels of biodegradability (Falzarano et al., 2024). In addition, most standards target unrealistic degradation rates in short periods of time, making it difficult to manage bioplastics waste under real treatment conditions. Therefore, a comprehensive understanding of the fundamentals of biological treatment technologies is necessary to design an appropriate end-of-life route for bioplastics.

In this context, the aim of this review is to provide a realistic overview of the current state of the art in the management of bioplastics. This review systematically addresses the fundamentals of anaerobic digestion and composting treatment of organic waste, with a particular focus on the biodegradation of plastics. It also discusses strategies to improve the performance of anaerobic digestion in the treatment of biodegradable plastics and their impact on the performance of biological treatment processes. Finally, the installed anaerobic digestion and composting treatment capacity and the main substrates used in different countries around the world are analyzed.

2. Microbiology of anaerobic digestion and composting of organic waste

2.1. Anaerobic digestion

Anaerobic digestion is a waste management method used worldwide to treat organic waste while recovering energy in the form of biogas (Bátori et al., 2018). Biogas is considered a green energy vector, mainly composed of 60–70% methane (CH_4), 30–40% carbon dioxide (CO_2), and other trace compounds such as hydrogen sulfide (H_2S), ammonia (NH_3), hydrogen (H_2) and nitrogen (N_2) (Rodero et al., 2018). Much research has been conducted to optimize and scale up this process, for which it is essential to understand its fundamentals. In this context, the following sections are dedicated to the description of the microbiology of the process and the description of the types of digesters.

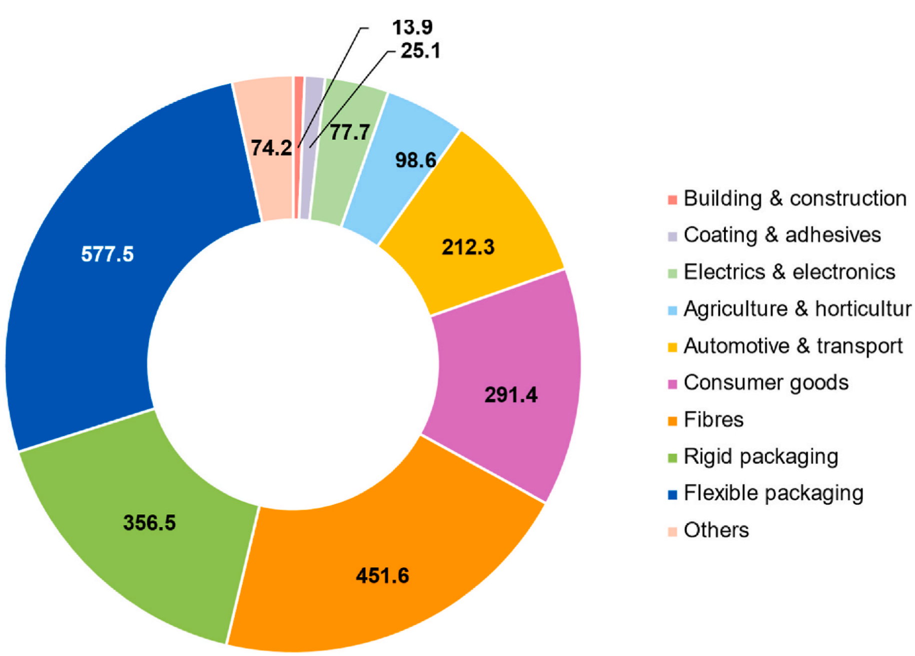


Fig. 1. End-use markets of plastics in 2023, data represents millions of tons (Plastics Europe, 2024).

2.1.1. Microbiology of the anaerobic digestion process

Anerobic digestion is a complex biological process in which organic matter is degraded by a consortium of different microorganisms to produce biogas and a nutrient-rich liquid digestate in the absence of oxygen (Vargas-Estrada et al., 2021). As shown in Fig. 2, the anaerobic digestion process can be divided into four main stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis which can be carried out in series or in parallel.

2.1.1.1. Hydrolysis stage. During substrate hydrolysis, complex organic compounds, such as lipids, proteins and carbohydrates, are depolymerized into simple molecules such as long-chain fatty acids, amino acids and sugars, which can be easily degraded. *Clostridia* spp. are involved in this stage as these microorganisms are able to produce cellulases, lipases, proteases and other enzymes necessary for hydrolysis (Peng et al., 2018). Bacteria of the genus *Clostridium, Peptostreptococcus* and *Bifidobacterium* are responsible for protein degradation, while lipids can be hydrolyzed and degraded to glycerol and various fatty acids by aerobic, facultative aerobic or strict anaerobic organisms.

2.1.1.2. Acidogenic stage. In the acidogenic stage, the soluble compounds produced in the hydrolysis stage are converted into short-chain fatty acids, commonly known as volatile fatty acids (VFAs), such as acetic acid, propionic acid, butyric acid, valeric acid, etc. Alcohols such as methanol and ethanol, aldehydes, CO_2, and molecular hydrogen (H_2) are also produced (Gunes et al., 2019). Acidogenic bacteria include species of the genera *Butyrivibrio*, *Propionibacterium, Clostridium, Bacteroides, Ruminococcus, Bifidobacterium, Lactobacillus, Streptococci, Endobacteria*, among others. The activity of some fermentative bacteria depends on the concentration of H_2 in the media, with a lower partial pressure of H_2 in the culture broth typically associated with a higher bacterial acidogenic activity (Gunes et al., 2019).

2.1.1.3. Acetogenic stage. The intermediates produced in the acidogenic stage (*i.e.* VFA, alcohols, H_2) are then converted into acetate, H_2 and CO_2 by acetogenic bacteria. Acetogenic bacteria can consume soluble organics for microbial anabolism to ultimately form acetic acid, H_2 and CO_2 via a proton-reducing acetogenic pathway or homoacetogenic pathway. Acetogenic metabolism depends on the presence of methanogenic archaea, as they are responsible for the consumption of acetate and H_2. Therefore, the syntropy between methanogens and acetogenic bacteria is essential to stabilize this stage. Acetate accumulation can lead to biogas inhibition (Gunes et al., 2019; Huang et al., 2020; Peng et al., 2018; Srisowmeya et al., 2020).

2.1.1.3.1. Methanogenic stage. During methanogenesis, acetic acid, H_2 and CO_2 are converted to biogas. CH_4 can be produced by acetoclastic archaea, which consume acetic acid, and hydrogenotrophic archaea, which consume H_2 and CO_2 to produce CH_4. Acetoclastic methanogens can produce nearly 70% of the total CH_4 (Yang et al., 2019); however, hydrogenotrophic methanogenesis may be the dominant pathway depending on substrate properties and environmental and operational conditions. Methanogenic archaea include species of the genera *Methanothrix, Methanobacterium, Methanococus, Methanobrevibacter, Methanogenium*, *Methanosarcina,* and other. *Methanosarcina* can use both pathways to produce CH_4 (Srisowmeya et al., 2020).

2.1.2. Anaerobic digester configurations

The anaerobic digestion process can be classified according to the number operating stages, the type of feeding and mixing, temperature, etc. (Liu et al., 2011). Fig. 3 shows the basic classification of the anaerobic digestion process.

Anaerobic digesters come in a variety of shapes, including cylindrical structures and covered lagoons. However, cylindrical digesters are the most common configuration. The biogas can be collected directly in the digester headspace or piped to an external gasometer. The inlet and outlet pipes are usually located on opposite sides of the digester, whilst sludge removal generally takes place at the bottom of the digester. Table 1 summarizes the main characteristics of different anaerobic digester configurations, highlighting the variety of design options available.

2.2. Composting

Composting is a widely adopted method for organic waste such as

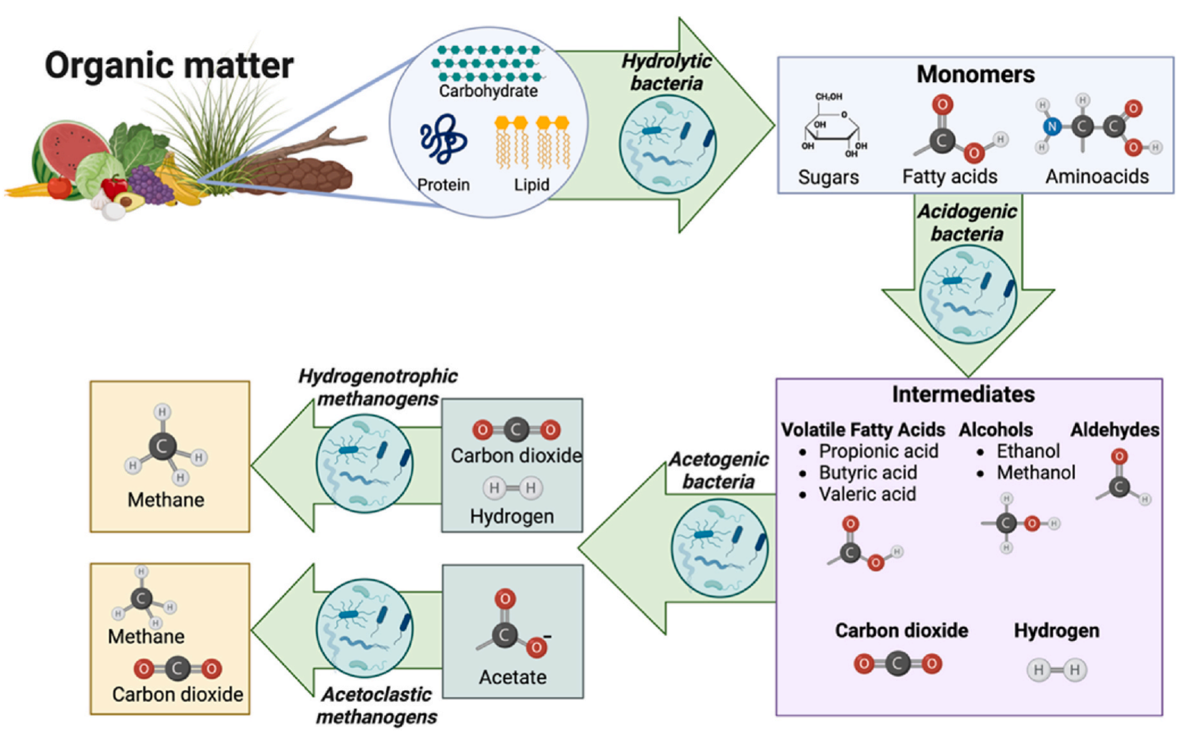


Fig. 2. Main compounds produced and bacteria involved in the anaerobic digestion process.

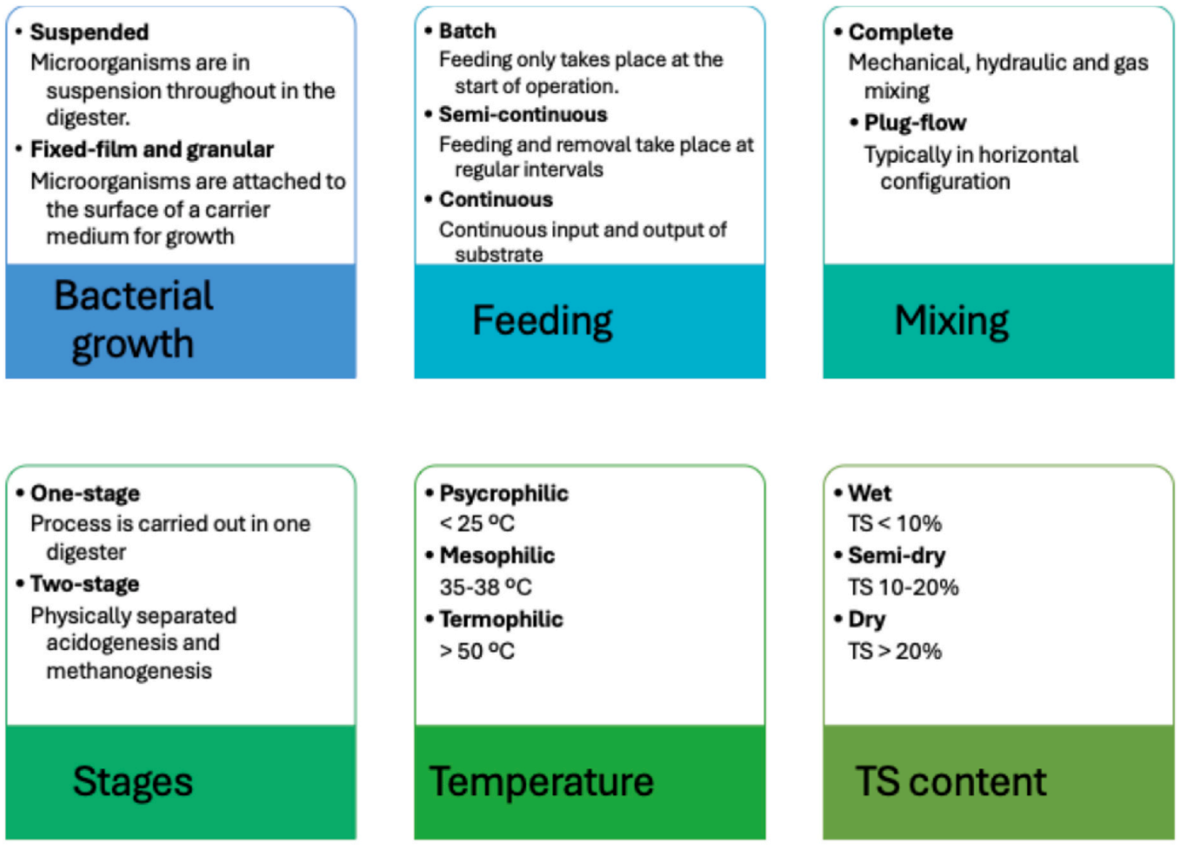


Fig. 3. Classification of the anaerobic digestion process according to the operational conditions. Adapted from Kirchmeyer et al. (2020) and Liu et al. (2011). TS: total solids.

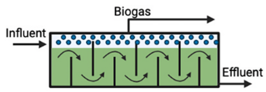
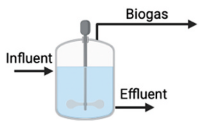
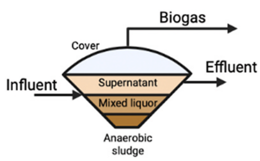
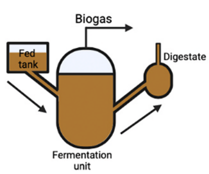
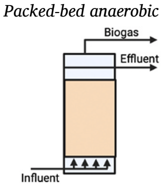
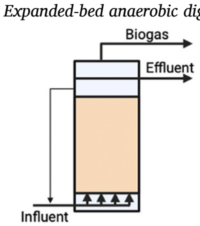
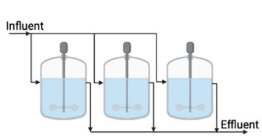
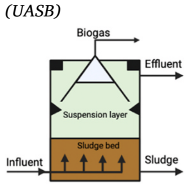
food waste disposal and animal manure management and is typically carried out in open-air windrows or in-vessel systems (Ebenezer et al., 2020; Liu et al., 2020). The attractiveness of this technique lies in the nutrient-rich compost produced, which can be used as a fertilizer or a soil conditioner, act as a natural pesticide, and sequester carbon (Schmidt et al., 2019; van der Linden and Reichel, 2020). In fact, it is estimated that 1 ton of fresh compost can sequester between 60 and 150 kg of CO_2 (ECN, 2022a). Composting can prevent the landfilling and/or incineration of organic waste and it is typically associated with the treatment of agricultural waste from farms and gardens (Barik, 2018). Nevertheless, it is considered a slow process as it can take between 3 months and 2 years, depending on the composition of the organic waste and operating conditions (Ebenezer et al., 2020; Schmidt et al., 2019). Compared to home composting, the process can be accelerated under industrial composting conditions, producing compost in a matter of weeks due to to advanced control of operating conditions such as temperature, moisture, oxygen levels, and the carbon-to-nitrogen ratio. Composting can also be accelerated by shredding or turning the biomass, or by adding inoculants, natural additives and/or minerals (Ebenezer et al., 2020).

Recently, the European Composting Network (ECN) proposed the use of organic fertilizers as a viable and sustainable technique to maintain and increase soil organic carbon in mineral soils (ECN, 2022a). Composting is considered as a key platform to achieve a sustainable agriculture since the produced compost can supply nutrients to plants and improve the soil quality. Moreover, compost increases the organic carbon in soil, rectifies soil structure, improves water holding capacity, water infiltration rate and soil tilth (Ahmad et al., 2021). In this sense, the sustainability of the composting process relies on the fact that eroded soils can be restored by using organic waste. Other sustainable strategies such as the sustainable circular economy and climate-neutral models promote the recycling and revalorization of wastes, where composting could play a key role in achieving these goals (Idris et al., 2023). The following sections aim to describe the basics of composting. The general system designs are also reviewed and described.

2.2.1. *Microbiology of composting*

The composting process can be divided into four stages: mesophilic, thermophilic, cooling and maturation. The first three stages involve the bio-oxidation or degradation of the organic matter (Schmidt et al., 2019) (Fig. 4). During the degradation phase, oxygen is required to break down the organic matter into CO_2, H_2O and NH_3, while the maturation phase stabilizes the organic matter and produces compost. Thus, during the process of composting, raw complex organic materials are converted into more stable compounds (compost) such as humic substances, whilst easy degradable compounds are mineralized to CO_2, NH_3 and H_2O. Humic substances include humic acids, fulvic acids and humins, and the presence of these substances give compost the potential to improve soil structure and enhance plant growth (Guo et al., 2019; Liu et al., 2023). Humic substances can be formed by two pathways: 1) the lignin pathway; and 2) the polyphenol pathway. In the lignin pathway, the partial degradation of lignin results in phenolic and quinone species that serve as precursors of humic substances. In the polyphenol pathway, small molecules including polysaccharides and proteins, are condensate to form humic substances. Typically, humic substances precursors are formed during the heating and thermophilic phases, while the humic substances are mainly formed during the cooling and mature phases. Humic substances can serve as indicators for evaluating the composting process, i.e. high concentration of fulvic acids indicates low maturity and humification of the compost whereas a mature compost has a high

Table 1
Configurations and characteristics of anaerobic digesters.

Digester type	Characteristics	Reference
Baffled anaerobic digester 	Internal baffles to direct the flow. Plug flow design. Low tolerance to shock loads. Recirculation of effluent can prevent inhibition by shock loads.	FAO (2011)
Continuously stirred tank reactor (CSTR) 	One or more agitators ensure that the substrate is continuously mixed with the microorganisms Maintains a uniform concentration of substrate and microorganisms. The substrate concentration is the same throughout the digester volume and in the effluent.	(Fallis, 2013; FAO, 2011; Kirchmeyer et al., 2020)
Covered lagoon 	Typically used for animal and municipal wastewater treatment in small communities. Typically buried in the ground. Operates at ambient temperature for HRTs of 50–70 days. Cost-effective and easy to operate. Require large volumes and surface areas.	Fallis (2013)
Household digesters 	Most household digesters are located in Asia. Cheap to build and easy to maintain and operate. The most common are the "Chinese" and the "Indian" configurations. Chinese is usually built underground. Indian has a floating digester model.	Rajendran et al. (2012)
Packed-bed anaerobic digester 	Also known as anaerobic biofilters. Usually packed with plastic or ceramic rings to foster bacterial attachment. Bacteria can accumulate in the packing material leading to clogging. Backwashing is required to prevent clogging. Effective for treating low strength wastewater.	Fallis (2013)
Expanded-bed anaerobic digester 	It is an improvement of the packed-bed anaerobic digester. The media bed is expanded to allow movement inside the digester. The mixed liquor is recirculated. Plastic rings, granular activated carbon and sand are used as support. Low HRT but expensive to build and operate.	Fallis (2013)
Plug-flow reactor (PFR) 	Typically configured as horizontal and elongated tanks or tubes. Vertical configurations are also possible. Can be scaled up for wet and dry digestion. Substrate concentration is highest at the inlet and decreases throughout the digester.	(Fallis, 2013; Kirchmeyer et al., 2020)
Sequencing batch reactors 	Consist of multiple batch reactors interconnected in parallel or series. High bacterial concentration resulting in high digestion efficiency. Semi-continuous feeding. Complex operation and control.	Fallis (2013)
Up-flow anaerobic sludge blanket digesters (UASB) 	Widely used for the treatment of organic wastewater. Influent enters at the bottom and moves through the sludge bed or an anaerobic granular suspension. Cost effective design.	Liu et al. (2011)

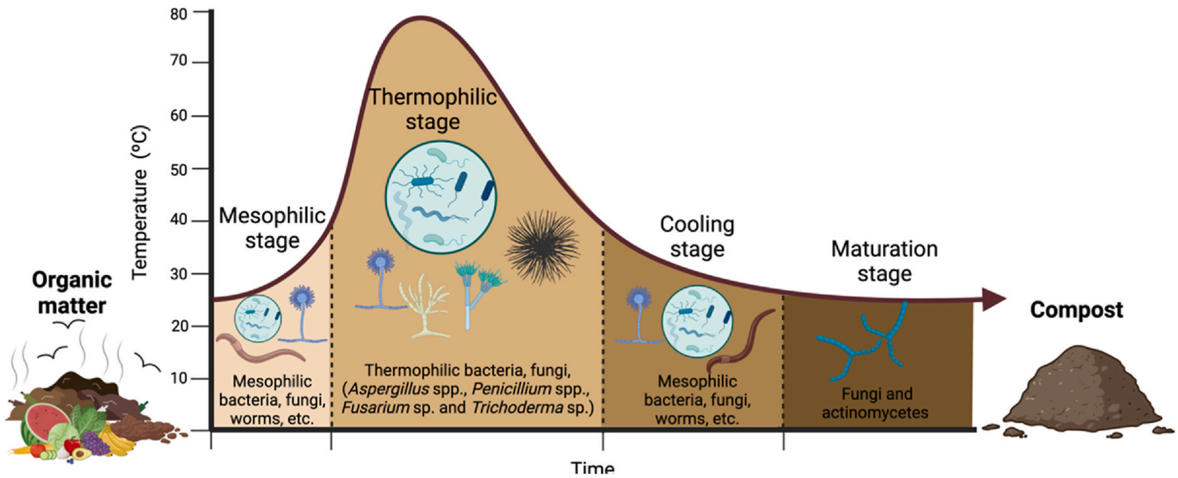


Fig. 4. Main microorganisms involved and temperature ranges during the different stages of composting.

concentration of humic acids (Guo et al., 2019). In this sense, the composting process is stopped when the compost is mature and has a high concentration of humic acids. Additionally, particles are reduced to "crumb" size, the pile is reduced to 50% of the initial size, the temperature is stabilized to ambient temperature, and the color of the compost is dark brown to black (Bernal et al., 2009). Additionally, there are no visible organic residues, and the compost emits a pleasant earthy odor, and has a crumbly, soil-like texture. One part of the organic matter present in the compost is consumed by the microorganisms to grow and is commonly referred as biomass (Soni and Devi, 2022).

2.2.1.1. Mesophilic stage. This initial phase typically takes place at 25–40 °C, where compounds such as sugars and proteins are easily degraded by primary decomposers *i.e.* actinobacteria, bacteria and fungi. Microorganisms of the genera Alphaproteobacteria such as *Afipia* and *Hyphomicrobium,* Gammaproteobacteria such as *Rhodanobacter,* the Actinobacteria *Intrasporangium* and Firmicutes such as *Bacillus* have been identified in mixed waste compost. Additionally, thermophilic bacteria such as *Candida ethanolica, Bacillus cereus* and *Alcaligenes faecalis* have been identified in municipal solid waste compositing, while the thermophilic bacteria *Pseudomonas aeruginosa, P. putida, Sphingobacterium moltivorum, Delftia tsuruhatensis, Stenotrophomonas humi, Ochrobactrum oryzar, Micrococcus luteus* and others have been identified in landfill compost. Other organisms such as millipedes, mites, worms and other mesofauna can act as catalysts during composting. (Schmidt et al., 2019; Soni and Devi, 2022).

2.2.1.2. Thermophilic stage. The thermophilic stage is carried out at 55–65 °C. In this stage, most of the mesophilic organisms die and are degraded along with the remaining organic matter by thermophilic organisms. The fungal groups identified during the composting of municipal waste were *Aspergillus* spp., *Penicillium* spp., *Fusarium* sp. and

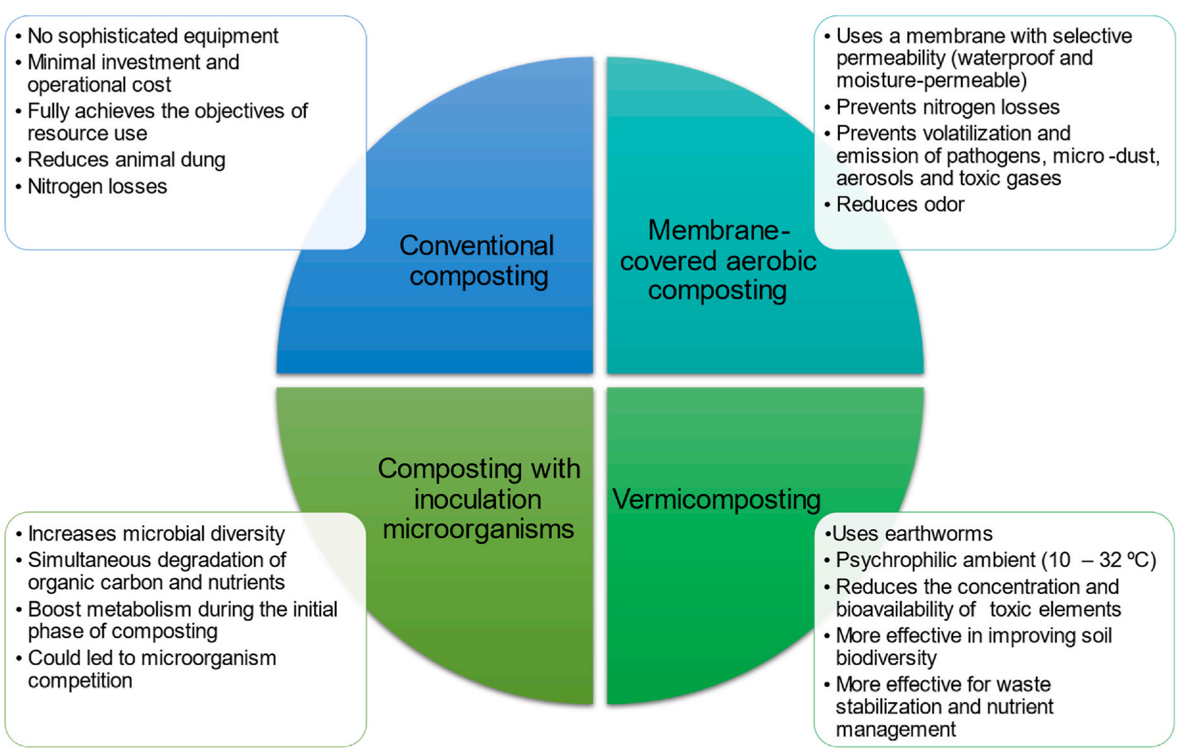


Fig. 5. Classification of composting processes (Xu et al., 2023).

Trichoderma sp., whereas *Trichophyton ajelloi, Aphanoascusreticulisporus, A. fulvescens, A. durus, Arthroderma quadrifidum, Chrysosporium anamorph of Arthroderma curreyi, Myceliophthoravellerea, C. keratinophilum* with its teleomorph *A. keratinophilus, C. europae, C. tropicum,* and *Microsporum gypseum* have been identified in sewage and municipal waste compost. This specific stage is recognized as a sanitization stage, where pathogens such as larvae, weed seeds and phytotoxic compounds are destroyed mainly by the increase in temperature. This stage can last several days or weeks. (Schmidt et al., 2019; Soni and Devi, 2022).

2.2.1.3. Cooling stage. The cooling stage is also referred to as the second mesophilic stage, where the temperature decreases as the thermophilic organisms consume the organic substrates. During the cooling stage, the degradation of starch and cellulose is carried out by mesophilic organisms. The main microorganisms present in this stage are bacteria and fungi, but macroorganisms such as earthworms and sowbugs are also present (Soni and Devi, 2022). In this stage, the previously produced amino acids, peptides and polyphenolic compounds polymerized to form humic acids (Wang et al., 2023).

2.2.1.4. Maturation phase. During this phase, the fungal population increases and the bacterial communities decrease. The main microorganisms present are fungi and actinomycetes such as *Nocardiopsis lucentensis* and *Saccharomonospora azurea*, which break down complex compounds such as cellulose and lignin. Other organisms such as earthworms, insects and woodlice may also be present. This process can take several months to a year (Schmidt et al., 2019; Soni and Devi, 2022).

Table 2
Classification of composting technologies.

System	Type of agitation	Characteristics	Reference
Passively aerated windrows	Static	• It has a ventilation layer made of materials (wood chips, wheat straw) with good air permeability. • It relies on convective air to provide oxygen, which ultimately controls the temperature. • Typically, PVC pipes are placed at the bottom of the compost. • Cost effective and does not require skilled labor. • Most economical aeration method.	(Ghassan Alsultan et al., 2023; Liu et al., 2022)
Forced aeration	Static	• Blowers are used to force (positive) or extract (negative) the air at a specific rate and velocity. • Negative aeration can be used with the collected exhaust air. • Odors can be controlled. • Reduced maturation time. • High capital cost.	(Ghassan Alsultan et al., 2023; Lin et al., 2019)
In-vessel	Static/ agitated	• Organic waste is placed in closed systems where airflow and temperature can be well controlled. • Occupies a small footprint and the process is not affected by external factors. • High capital, operating and maintenance costs.	(Lin et al., 2019; Liu et al., 2022; Schmidt et al., 2019)
Windrow	Agitated	• Organic matter is formed into windrows and frequently turned/agitated to introduce oxygen. • Simple to operate. • Longer composting time. • Climate sensitive.	(Lin et al., 2019; Liu et al., 2022; Schmidt et al., 2019)

2.2.2. Composting systems configurations

Composting systems have progressed significatively in recent years and can be divided into conventional and unconventional methods (Fig. 5). Unconventional methods include membrane-covered aerobic composting, vermicomposting and composting with inoculated microorganisms (Xu et al., 2023). In addition, depending on the technology used, aeration can be passive, forced or by turning. Thus, composting can be classified as shown in Table 2.

3. Anaerobic and aerobic biodegradability of bioplastics

Bioplastic manufacturing is a fast-growing industry with a positive social acceptance (Cucina et al., 2021a; Dilkes-Hoffman et al., 2019). Global bioplastic production capacity is forecasted to increase from 2.2 million tons in 2023 to 6.3 million tons by 2027 (European Bioplastics, 2023). Fig. 6 shows the global production of bioplastics in 2023, by the type of material. Bioplastics are defined as a plastic that is biobased and/or biodegradable (Dilkes-Hoffman et al., 2019), and their biodegradability depends on, not only on the surrounding environment, but also on their chemical and physical properties such as surface area, surface type, molecular weight, chemical structure, crystallinity, melting temperature, among others (Tokiwa et al., 2009). A proper classification, separation and collection is essential to ensure a sustainable end of life treatment for bioplastics. Different treatment techniques are available to treat bioplastics, however, biological methods are considered the most proper option to manage biodegradable plastics. During biodegradation, bioplastics polymers are degraded to their most simple monomer by microbial microorganisms via metabolic or enzymatic pathway. Then, the monomer molecules are degraded into CO_2, water, biomass and minerals without generating any toxic subproduct. However, bioplastics degradation rate depends on the activity of the microbial community and on environmental parameters with temperature and moisture being critical. Additionally, other factors such as particle size, high molecular weight, addition of additives, etc., significantly affect the biodegradability of bioplastics and can slow-down the process (Nayanathara Thathsarani Pilapitiya and Ratnayake, 2024; Rasheed et al., 2024).

During plastic biodegradation, the hydrolysis of polymers has been identified as the key stage to fulfill biodegradation. This primary stage is conducted by enzymes segregated by different microorganisms, i.e. bacteria or fungi, which are typically adhered to the polymer surface and cause surface erosion. During this primary stage, oligomers with lower molecular weight are obtained which are easier to assimilate by microorganisms, and are converted to CO_2, CH_4, water and biomass. The main microorganisms involved during bioplastics biodegradation are summarized in Table 3.

The biodegradation of bioplastics can follow two pathways, aerobic and anaerobic, which can be implemented in two different types of biological treatment: composting and anaerobic digestion, respectively (Bátori et al., 2018). The following sections are devoted to comprehensively explain the anaerobic digestion and aerobic composting of bioplastics.

3.1. Anaerobic biodegradation of bioplastics

Some biodegradable plastics can be converted to CO_2 and/or CH_4, water, salts and biomass under anaerobic conditions (Yu et al., 2023). Anaerobic digestion of bioplastics provides a direct route to energy recovery from bioplastics, as the biogas produced is easy to collect and use. The anaerobic digestion of bioplastics can be carried out under mesophilic conditions (35–37 °C), which does not require an intensive energy input compared to composting. Additionally, the interest in anaerobic digestion of bioplastics is based on the fact that these materials are

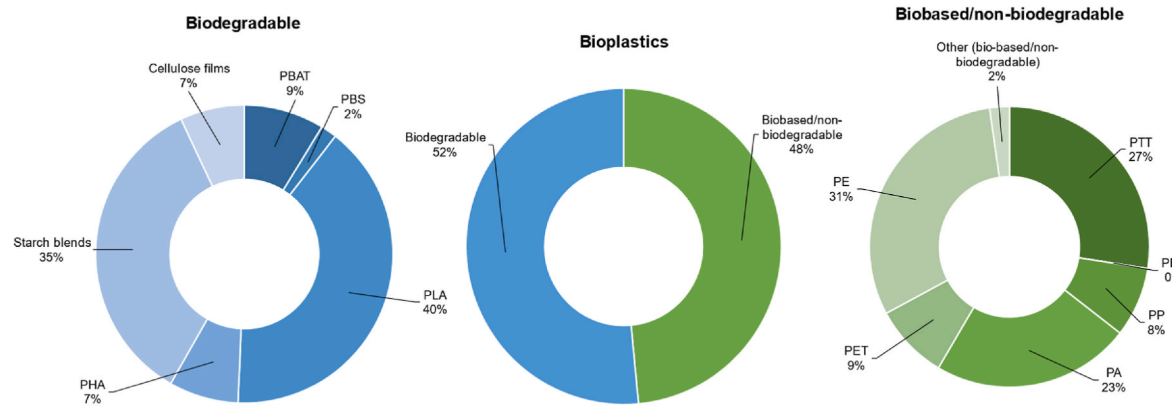


Fig. 6. Global production of bioplastics in 2023, taken from (European Bioplastics, 2023).

Table 3
Main microorganisms identified during bioplastics biodegradation (Cazaudehore et al., 2022b, 2023; Peng et al., 2022; Pooja et al., 2023; Yagi et al., 2014; Zaborowska et al., 2023).

Bioplastic	Monomer	Chemical caracteristics	Biodegradation process	Microorganisms involved in anaerobic digestion	Microorganisms involved in composting
PLA	Lactide	The cleavage of esther bonds facilitates PLA degradation	PLA is hydrolyzed and then decomposed by microorganisms into CO_2 and water.	*Chloroflexi, Actinobacteria, Treponema, Paludibacter, Rubrobacter* and *Leptolinea* *Bacteroides, Terrimonas, Cytophaga, Desulfofaba, Curvibacter* and *Thermomonas, Tepidimicrobium* sp. and *Moorella* sp	*Saccharothrix, Kibdelosporangium, Pseudonocardia, Lentzea,* and *Amycolatopsi.*
PBAT	Copolymer of adipic acid, 1,4-butanediol, and terephthalic acid	The aliphatic co-aromatic polyesters are difficult to degrade	Biodegradation is possible by enzymes including carboxylic ester hydrolases, including carboxylesterase, arylesterase, triacylglycerol lipase, and cutinases	*Bacteroidota, Chloroflexi, Desulbobacterota, Firmicutes, Euryarchaeots, Acidobacteriota, Coprothermobacterota, Proteobacteria.*	Bacteria: *Thermobifida fusca, Pelosinus fermentans, Clostridium botulinum, Pseudomonas* sp.
PCL	ε-caprolactone	PLC is used as a carbon and energy source by microorganisms	Biodegradation is possible by cutinolytic enzymes.	*Bacteroidota, Chloroflexi, Desulfobacterota, Firmicutes, Euyarchaeota, Proteobacyeria, Coprothermobacterota.*	Bacteria: *Clostridium* sp. Fungi: *Aureobasidium* sp., *Cryptococcus* sp., *Aspergillus flavus, A. niger, A. fumigatus, Chaetomium globosum, Pencillium funiculosum, Fusarium* sp. *Clonostachys rosea, Trichoderma* sp.
PHAs	Hydroxycarboxylic acids	Produced by bacteria under stress conditions. More than 150 types of PHAs have been identified	Polymer bonds are hydrolyzed into oligomers. Then, PHAs are converted into trimer and dimer units to be finally degraded.	*Arcobacter thereius, Clostridium* sp, *Enterobacter* sp., *Cupriavidus* sp., *Peptococcaceae bacterium Ri 50, Bacteroides plebeius* and *Catenibacterium mitsuokai,*	Bacteria: *Enterobacter* sp, *Bacillus* sp, *Gracilibacillus* sp., *Pseudomonas lemoigne, Comamonas* sp. *Acidovorax faecalis, Aspergillus fumigatus, Variovorax paradoxus, Streptomyces* sp, *Aspergillus* sp. Fungi: *Penicillium pinophilium, Penicillium funiculosum, Paecilomyces lilacinus, Aspergillus fumigatus, Emericellopsis minima*
Starch-based	Glucose monomers with α 1,4 linkages	Highly degradable. Linear and branched structures	The glycosidic bonds are hydrolyzed into oligomers, then the α-glycosidic linkages are hydrolyzed by fungi and bacteria. Finally, glucose bonds are broken by oxidative cleavage.	*Clostridium, Treponema* and *Paludibacteraceae, Limnochordia, Clostridium thermarumm*	Bacteria: *Bacillus* sp., *B. amyloliquefaciens, B. subtilis, B. licheniformis, B. stearothermophilus, B. megaterium, B. circularis.* Fungi: *Clonostachys rosea, Trichoderma* sp.

mixed with the organic fraction of municipal solid waste (OFMSW) and their concentration in the bio-waste stream is expected to reach high values (up to 10% of total organic matter) in the next years (Cucina et al., 2021a).

As previously mentioned, anaerobic digestion of biodegradable bioplastics is an attractive end-of-life option. However, the chemical composition of these materials must be carefully considered, as it significantly influences their suitability for anaerobic digestion. For instance, polyhydroxyalkanoate (PHA) blends and sugarcane cellulosic fiber-based plastics exhibit a good biodegradability during anaerobic digestion, degrading faster than other plastics and similarly to OFMSW (Battista et al., 2021; Yagi et al., 2014). On the other hand, poly(lactic acid) (PLA) and starch blends have shown slower biodegradability and longer retention times compared to PHAs under anaerobic digestion conditions (Battista et al., 2021; Cucina et al., 2021a, 2021b; García-Depraect et al., 2022). Although their anaerobic biodegradability depends on the environmental parameters such as temperature, it can be enhanced by pretreatments such as alkaline and thermo-alkaline pretreatments.

Anaerobic digestion of bioplastics faces several significant challenges that need to be overcome for full-scale implementation. The primary challenge is the inherently slow biodegradation rate of many bioplastics, which hinders efficient anaerobic digestion processes. Nonetheless, this limitation can be addressed by different techniques such as bioplastic pretreatment, co-digestion, acclimatation or bioaugmentation with specific bacteria to anaerobically degrade the target bioplastics (Fig. 7).

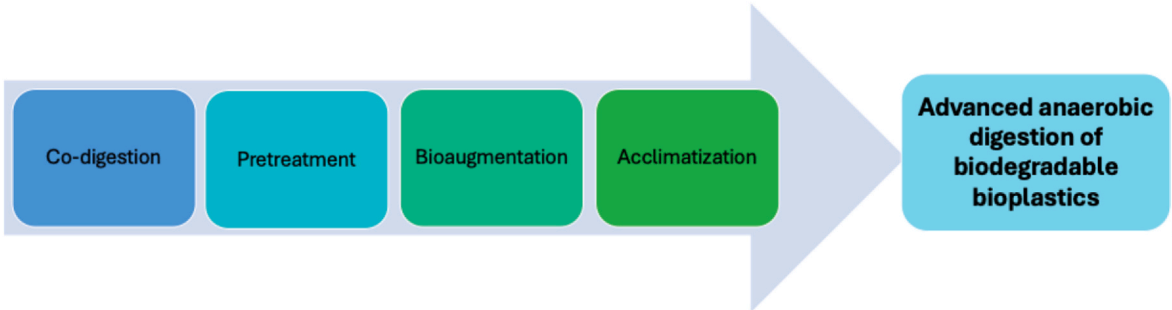


Fig. 7. Strategies to improved anaerobic digestion of bioplastics.

Pretreatment techniques for bioplastics have recently been fully reviewed by (Mat Yasin et al., 2022) and successfully tested by the Institute of Sustainable Processes in collaboration with Nestle (Garcia-Depraect et al., 2023). In the following section, this review will focus on the impact of pretreatments on the anaerobic co-digestion of bioplastics and organic waste.

3.1.1. Microbial communities involved in anaerobic digestion of bioplastics

Anaerobic digestion is widely recognized as a biological process involving a large biodiversity of anaerobic bacteria, facultative bacteria and archaea. Indeed, a better understanding of these populations is fundamental for improving the anaerobic digestion of bioplastics (Cazaudehore et al., 2023). Although the literature investigating the microbial populations involved in the anaerobic digestion of bioplastics is scarce (Cazaudehore et al., 2022a), recent studies have identified a change in the microbial population structure during the anaerobic digestion of PLA, poly(3-hydroxybutyrate) (PHB), thermoplastic starch (TPS) and others. For instance, it has been shown that the relative abundance of *Chloroflexi, Actinobacteria, Treponema, Paludibacter, Rubrobacter* and *Leptolinea* increased during the thermophilic anaerobic digestion of PLA, which have been regarded to be key in PLA methanization (Zaborowska et al., 2023). Interestingly, when PLA was thermally and chemically (alkaline) pretreated, an increase in the microbial abundance of *Bacteroides, Terrimonas, Cytophaga, Desulfofaba, Curvibacter* and *Thermomonas,* was recorded (Zaborowska et al., 2023). This particular microbial communities have also been identified as responsible for methane production in digesters with high VFA concentrations (Guo et al., 2015; Zaborowska et al., 2023). This is in line with the findings recently reported by (Garcia-Depraect et al., 2023), where alkaline pretreated PLA showed no lag phase, indicating that the lactic acid released during PLA depolymerization was readily metabolized. On the other hand, the thermophilic anaerobic digestion of PLA supported the growth of *Tepidimicrobium* sp. and *Moorella* sp., which were previously reported to play an important role in the depolymerization of PLA (Cazaudehore et al., 2023; Tseng et al., 2019).

During the mesophilic anaerobic digestion of TPS, an increase in the relative abundance of *Clostridium, Treponema* and *Paludibacteraceae* was observed. Under thermophilic conditions, *Clostridium* was still dominant together with *Limnochordia* during TPS anaerobic digestion. More specifically, *Clostridium thermarumm,* whose ability to degrade starch has recently been demonstrated, was identified in the thermophilic broth (Cazaudehore et al., 2023).

On the other hand, the anaerobic biodegradation of PHB or PHA requires the synthesis and release of extracellular enzymes, such as PHA depolymerases and lipases (Venkiteshwaran et al., 2019). In this context, *Arcobacter thereius* has been identified as a key player in PHB degradation under mesophilic conditions (Cazaudehore et al., 2022a). Additionally, *Arcobacter thereius* and *Clostridium* sp. have also been identified during the anaerobic digestion of PHB (Yagi et al., 2014). In contrast, *Enterobacter* sp. and *Cupriavidus* were identified under thermophilic conditions, which are known for playing a key role in the anaerobic degradation of PHB (Cazaudehore et al., 2022b, 2023; Knoll et al., 2009). Similarly, *Peptococcaceae bacterium Ri 50, Bacteroides plebeius* and *Catenibacterium mitsuokai*, were identified during the thermophilic degradation of PHB (Yagi et al., 2013).

Other anaerobic bacteria have shown potential for bioplastic degradation. For instance, *Ilyobacter polytropus* and *Clostridium botulinum* were identified in the mesophilic degradation of 3-hydroxybutyrate and poly(3-hydroxybutyrate-*co*-3-hydroxyhexanoate) (PHBH) (Abraham et al., 2021). Similarly, the biodegradation of a blend of poly(butylene adipate-*co*-terephthalate) (PBAT)/PLA polymers under thermophilic conditions was attributed to *Brevundimonas* and *Sphingobacterium* (Peng et al., 2022).

Regarding methanogenic communities, different archaeal communities such as *Methanolinea*, Methanomassiliicoccus, Candidatus *Methanofastidiosum*, *Methanobacterium*, *Methanobrevibacter*, and *Methanosphaera* were found in a recent study on the anaerobic batch degradation of PHBH in co-digestion with different organic wastes (García-Depraect et al., 2024). *Methanobrevibacter* and *Methanosphaera* were the major archaeal communities associated with methane production during the thermophilic anaerobic digestion of rigid PLA and starch-based spoons (Bandini et al., 2022b). *Methanosaetaceae, Methanosarcinaceae* and *Bathyarchaeiai* were identified in the anaerobic degradation of PLA and PBAT (Kriswantoro et al., 2023). Peng et al. (2024) identified the hydrogenotrophic methanogen *Methanoculleus* in the mesophilic anaerobic degradation of PLA/PBAT of plastic bags. Interestingly the abundance of this particular archaea was >90%, suggesting that the main methanogenic pathway was hydrogenotrophic (Peng et al., 2024). On the other hand, under thermophilic conditions, the addition of bioplastics, PLA/PBAT-based decreased the abundance of *Methanosaeta* and *Methanolinea*; and the dominating archaeal communities were *Methanobacterium, Methannocculleus* and *Methanosarcina* (Peng et al., 2024). Candidatus Methanoplasma, *Methanoculleus*, *Methanosphaera* and *Methanothermobacter* have also been retrieved in the thermophilic anaerobic digestion of PLA glasses, however their abundance was <2% (Clagnan et al., 2023). *Methanothermobacter* has also been identified in the thermophilic anaerobic degradation of the blend of PBAT/PLA/starch at an abundance of 56% (Yu et al., 2023). Venkiteshwaran et al. (2019) observed that archaeal communities take longer to acclimatize compared to bacterial communities due to their sensitive nature. Therefore, research should focus on techniques to improve archaeal performance (Venkiteshwaran et al., 2019).

In this regard, more attention should be paid to the optimization of microbial populations in order to improve bioplastics depolymerization and further biomethanization. Thus, the combination of proper microorganisms, pretreatment and co-substrates could lead to a more efficient anaerobic digestion process that could be easily implemented in existing anaerobic digestion plants. Indeed, long-term studies are needed to assess the acclimation strategies of microbial communities during the anaerobic biodegradation of bioplastics.

3.2. Aerobic biodegradation of plastics

Composting has been considered a potential end-of-life management strategy only when bioplastics have a considerable amount of residual food, as mechanical recycling is not possible since existing recycling industries cannot deal with these contaminated plastic wastes (Ahsan et al., 2023). Compostable plastic is a type of plastic designed to decompose under controlled composting conditions of temperature (>55 °C) and moisture (60%) to produce water, CO_2 and compost. At this point it must be highlighted that compost production from bioplastic would be negligible based on the limited nutrient content of bioplastics. Biodegradation of bioplastics involves the breakdown of polymers into simple monomers, which can be used by microorganisms as a carbon or energy source. However, there is limited literature about the ecotoxicity and other impacts of bioplastics on microorganisms. Recently, De Bernardi et al. (2024) demonstrated that the presence of bio-packaging did not affect the bacterial communities during composting but the fungal communities significantly varied, and Mortierella, Mucor and Alternaria genera, which are capable of using bioplastics as a carbon source, were identified. Nevertheless, a post-composting ecotoxicity analysis conducted by the same authors, demonstrated that the presence of residual bioplastics in the compost had a limited DNA damage in earthworms and altered gut bacterial communities, which can significantly change compost fertility (De Bernardi et al., 2024). Genuinely, the presence of bioplastics is not desired in composting plants as it raises awareness due to potential risks to the process mainly caused by their slow biodegradation kinetics (Islam et al., 2024). For instance, Lavangnolo et al. (2020) demonstrated that during the composting of starch-based plastic bags at pilot scale, the size reduction took place in the thermophilic phase and the degradation occurred during the curing phase. Even though the monitored parameters indicated that the composting process of the waste matrix ended after 55 days, the biodegradation of the bioplastics did not meet the regulatory standards (size <2 mm) and 100 days were necessary to degrade the starch-based plastic bags (Lavagnolo et al., 2020). Additionally, bioplastics chemical composition also influences their biodegradation. For instance, PHAs biodegradation improves with increasing temperature, but the presence of long side-chains in their structure, such as in poly(3-hydroxybutyrate-co-3-hydroxyalkanoates) and poly(3-hydroxybutyrate-co-3-hydroxyhexanoate) lowers the degradation rate leading to incomplete biodegradation even under controlled industrialized processes (Li et al., 2007). Thus, the presence of bioplastic particles in the final compost, better known as micro and nanoplastics, are creating a particular concern due to their ecotoxicity and possible infiltration to the food chain. On the other hand, the quantity of bioplastics present in the compost also influences the process. For instance, the presence of 30% w/w of PLA bioplastics have demonstrated to lower the pH of the compost from 6.0 to 4.0, suppressing the microbial activity (Ghorpade et al., 2001). Thus, the type of bioplastic will benefit or affect the compost differently, depending on their chemical composition, size, quantity, etc. Thereby, special attention is necessary to properly identify and separate bioplastics that can undergo industrial composting without representing a risk to the environment.

Compostable plastics are typically made from renewable resources such as corn starch, potato starch, or other plant-based materials. To be labeled as compostable, they must meet specific standards (such as ASTM D6400 or EN 13432) that outline the time frame and conditions (including temperature and moisture) required for disintegration in a composting environment in industrial composting facilities. Some biodegradable plastics such as PLA, PHA, starch-based, poly(butylene succinate) (PBS), poly(ethylene succinate) (PES) and polycaprolactone (PCL) have been reported to be suitable for aerobic degradation in composting (Emadian et al., 2017). Some blends of PHA, PLA and starch-based have registered higher kinetic constants of degradation when submitted to thermophilic composting compared to anaerobic digestion (Cucina et al., 2021a). The degradation of bioplastics by composting has been recently reviewed by (Cucina et al., 2021a; Ahsan et al., 2023), and some general characteristics are summarized in Table 4.

It is important to address that the chemical composition of the aforementioned bioplastics makes them suitable for composting. For instance, starch-based bioplastics are composed of glucose monomers joined in α 1,4 linkages. On the other hand, PLA is formed by the condensation of lactic acid, whilst PHA are more complex polymers and are composed of hydroxycarboxylic acids. In this context, the degradation of bioplastics to their monomer does not represent a risk to the compost from a chemical point of view. However, little is known on the effects of bioplastics toxicity to microbial communities. Other bioplastic characteristics, such as thickness, presence of pollutants (*i.e.* presence of additives or fillers or food waste), directly affect the biodegradability of plastics (Falzarano et al., 2024; Gadaleta et al., 2023). For instance (Ruggero et al., 2020), reported a 90% biodegradation of a starch-based plastic bag in 20 days under thermophilic composting conditions (58 °C), while (Bandini et al., 2022a) reported a biodegradation ≤65% of a starch-based spoon in 22 days at 65 °C. Similar results were reported by (Bandini et al., 2022a) for the aerobic biodegradation of PLA spoons under composting conditions (65 °C), where only 65% of the total material was biodegraded in 22 days. Falzanaro et al. (2024) estimated that the complete biodegradation of PLA and Mater-Bi® cups by industrial thermophilic (58 °C) composting requires 130 and 180 days, respectively (Falzarano et al., 2024). Thus, even if bioplastics have a similar

Table 4
General conditions of the aerobic biodegradation of bioplastics by composting.

Bioplastic	Environmental parameters	Biodegradation	Produced monomer	Material (% of biodegradation)	Average biodegradation time (days)	References
Cellulose-based	Temperatures above 60 °C.	Slow biodegradation due to glycosidic bonds.	Cellobiose	Sponge cloths (100%)	154	(Ahsan et al., 2023; Vaverková and Adamcová, 2015)
PHA	Low temperature, low pH. Can be degraded in home composting	Slow biodegradation	Breaking polymer into oligomers and then into trimer and dimer units.	Ground plastic (30%)	60	(Ahsan et al., 2023; Sun et al., 2021)
PLA	High temperature humid environments. Mainly industrial scale.	Two stages: 1) Hydrolysis into monomers or oligomers. 2) Metabolization by microorganisms	Lactide monomer	Spoon (60%)	60	(Ahsan et al., 2023; Ruggero et al., 2022)
Starch-based	pH 7.0–8.0, 50% moisture. Industrial and home composting.	Directly degraded by microorganisms	Glucose monomer	Plastic bag (90–100%) Spoon (65%)	20–30 22	(Ahsan et al., 2023; Bandini et al., 2022a; Ruggero et al., 2020, 2022)

chemical composition, their physicochemical characteristics, such as material thickness, directly influence the biodegradation process. It is important to highlight that chemicals added to bioplastics, including fillers, flame retardants, antioxidants, stabilizers, plasticizers, pigments and trace elements could represent a high risk to living organisms and an environmental threat (Crema et al., 2024). For instance, bisphenol A and phthalates are toxic for humans and their use has been banned in the EU. The effect of chemical additives present in bioplastics is out of the scope of the current study but has been reviewed by (Hahladakis et al., 2018).

On the other hand, operational conditions such as temperature and moisture content have been reported as key operational parameters influencing plastic biodegradation. For instance (Ruggero et al., 2020), observed that PBAT degradation reached 17% at 45% moisture content and 58 °C for 5 days. Interestingly, the degradation of PBAT increased up to 80% when the moisture content increased to 55%, and finally a degradation of 90% of PBAT was reached at a moisture content of 45% under an extended thermophilic phase of 20 days. The authors concluded that moisture content was a factor that directly influenced microbial activity during PBAT degradation, and that both temperature and moisture must be in synergy to achieve satisfactory degradation of PBAT.

Indeed, although many bioplastics are labeled as 'compostable', their management requires special attention, as incomplete biodegradation of bioplastics limits the use of compost on agricultural land, as it does not meet the European Regulation (3 g kg^{-1} of impurities <2 mm) (Regulation (EU) 2019/1009) (Regulation (EU) 2019/1009, 2019).For instance, it has been reported that some cellulose-based bioplastics barely reached 20% of biodegradation after 22 weeks of composting at 58 °C (Vaverková and Adamcová, 2015). Moreover (Ruggero et al., 2022) estimated that rigid PLA bioplastics can reach the 90% of degradation in 2–3 years. Therefore, the search for strategies to enhance the aerobic degradation of bioplastics during composting is necessary.

Recently, some investigations have focused on the addition of fillers to bioplastics to improve their biodegradability, (Kalita et al., 2021) reported that the addition of 5% (wt) of algae extract to PLA increased the biodegradation of this particular material due to the high nitrogen content in the algae biomass, which mediated an accelerated biodegradation. On the other hand (Xing et al., 2023), reported that the addition of iron oxides (magnetite) increased the biodegradation of polyethylene (PE). The effect of magnetite was mainly driven due to the production of OH radicals resulting in higher oxygen-containing structures (Xing et al., 2023). Although the addition of fillers or the addition of iron oxides are promising, further research is still required to elucidate the effect of these additions on the materials properties, lifetime, interactions with food, etc. Finally, other strategies such as the addition of specific microorganisms capable of degrading the target bioplastics need to be further explored.

3.2.1. Bacterial communities involved in the composting of bioplastics

Microbial communities are an important factor that directly affects the biodegradation of bioplastics during composting. Although the disintegration and biodegradation of compostable bioplastics is influenced by operating parameters such as temperature, moisture and pH, microbial communities are ultimately responsible for degrading bioplastics into useful soil-like compounds. However, existing literature on the impact of bioplastics on microbial communities is scarce. Some studies have identified specific microorganisms, such as bacteria and fungi, as being primarily responsible for the biodegradation of bioplastics. For instance (Bandini et al., 2020), studied the microbial communities of aerobic composting of food waste and two bioplastics, PLA and starch-based bioplastic (SBB). Interestingly, the authors reported significant differences in the bacterial and fungal communities in the PLA assays. More specifically, the genus *Geobacillus* was identified, which has previously been shown to degrade PLA under composting conditions. For instance (Bandini et al., 2022b), identified the abundance of *Geobacillus thermodenitrificans* in the composting of rigid PLA spoons and food waste. Additionally, other bacteria such as *Bulkholderia cepacian, Geobacillus thermocatenulatus* and strains of *Pseudomonas* have also been identified as PLA degraders. Moreover, Actinomycetes have been also identified as PLA degraders (Kawai, 2010). On the other hand, fungal communities such as *Cladosporium sphaerospermum, Cladosporium cladosporioides*, *Penicillium chrysogenum* and *Penicillium roqueforti* have been identified as PLA degraders (Bandini et al., 2020, 2022b). More specifically, the *Cladisporium* genus produces cellulolytic and xylanolytic enzymes (Bandini et al., 2022b). Other enzymes, such as proteases and lipases have been identified as contributing to PLA degradation (Kawai, 2010). Bacteria such as *Caldicoprobacter* and *Firmicutes* have been identified in the composting of cellulose-based bioplastics (Bandini et al., 2020). Recently, Lu et al. (2023) reported that the composting of PLA + PBAT + bioplastic bag containing 20% of starch likely supported the abundance of Firmicutes, Proteobacteria and Bacteroidetes, resulting in an enhanced degradation of the bioplastics (Lu et al., 2023). In another study to evaluate the addition of mature compost in the composting of PBAT, *Thermobifida*, *Ureibacillus* and *Bacillus* were found to be major species in mature compost that could help in the biodegradation of PBAT (Wang et al., 2024).

Fungal communities such as *Mucor racemosus* were identified in the composting of rigid SBB spoons (Bandini et al., 2022b). Sun et al. (2021) reported an increase in *Firmicutes*, *Proteobacteria* and *Bacteroidetes* during the composting of PHA (Sun et al., 2021). Indeed, *Bacteroidetes* can degrade polymers, and an increase in their abundance may enhance the degradation of bioplastics. Fungal communities such as *Klebsiella pneumoniae,* which can degrade lignin, were present during the composting of cellulose-based and food waste (Bandini et al., 2020). Moreover, as *Ureibacillus thermosphaericus*, which can produce the enzymes catalase, esterase and amino acid dehydrogenase, is known to degrade lignocellulosic biomass (Bandini et al., 2020). In this regard, the isolation and enrichment of specific bioplastics degraders and the implementation of bioaugmentation strategies during composting could enhance the efficiency of bioplastics composting and thus reduce the risk of contaminated compost.

4. Anaerobic digestion and composting of bioplastics

4.1. Anaerobic digestion

Recently, Gadaleta et al. (2023) studied the mesophilic anaerobic digestion of pure cellulose acetate (CA) and a composite of cellulose acetate layered doubled hydroxide (with 5% wt. of LDH) (CA-LDH) in a full-scale industrial plant. The authors reported that only 36 and 50% of CA and CA-LDH, respectively, were biodegraded after 3–4 weeks (Gadaleta et al., 2023). Similarly, Cucina et al. (2022) reported that starch-based shoppers and PLA-based cutlery/dishes achieved only 30 and 21–28% degradation, respectively, with combined anaerobic digestion and composting (Cucina et al., 2021b). However, there is limited information available in this particular field of research, as specialized microbial communities may eventually develop in continuous anaerobic digesters, which would foster a complete anaerobic bioplastic degradation.

4.1.1. Anaerobic co-digestion of bioplastics

In the last years, researchers have focused on the biodegradability of bioplastics alone at laboratory scale. However, greater attention must be given to the anaerobic co-digestion of bioplastics with OFMSW, food waste (FW), sludge, and other co-substrates. This is crucial because co-digestion reflects the real-world conditions of bioplastic management, where bioplastics are often not separated from the food they contained or are processed within existing centralized waste treatment facilities in urban areas (García-Depraect et al., 2024). Interestingly, the anaerobic co-digestion of bioplastics such as PHA, PLA and PLA mixtures have been reported to increase the CH_4 yield when co-digested with OFMSW and food waste (Cucina et al., 2021b; Kang et al., 2022; Yu et al., 2023).

Table 5

Methane yield (mL_{CH4} gVS^{-1}) during the anaerobic co-digestion of bioplastics with different co-substrates. The increase in CH_4 yield is compared to the mono-digestion of the corresponding *bioplastic or **co-substrate. FW: food waste; OFMSW: organic fraction of municipal solid waste; SS: sewage sludge; SM; swine manure; N.A. not available data.

Bioplastic	Co-substrate	Temperature	Time (d)	CH_4 (mL gVS^{-1})	CH_4 increment (%)	% Biodegradation	Reference
PBAT	FW	Mesophilic	35	17.8	5**	N.A.	Yu et al. (2023)
PBAT/PLA/starch	FW	Mesophilic	35	21.1	25**	N.A.	Yu et al. (2023)
PHA	FW	Mesophilic	60	246[b]	159*	49.1	Kang et al. (2022)
PHB	FW	Mesophilic	85–97	308–398	–	~70	García-Depraect et al. (2024)
PHB	SM	Mesophilic	112	564	10*	~70	García-Depraect et al. (2024)
PHB	SS	Mesophilic	75	277	–	~70	García-Depraect et al. (2024)
PLA	FW	Mesophilic	35	19.1	15**	N.A.	Yu et al. (2023)
PLA	FW	Mesophilic	60	157–179[b]	1355-1542*	6	Kang et al. (2022)
PLA/starch[a]	OFMSW	Mesophilic	60	98	−43**		Cucina et al. (2021b)
Polyethylene	FW	Mesophilic	35	14.2	−11**	N.A.	Yu et al. (2023)
PBAT	FW	Thermophilic	35	19.5	14**	N.A.	Yu et al. (2023)
PBAT/PLA/starch	FW	Thermophilic	35	23.4	25**	N.A.	Yu et al. (2023)
PHA	FW	Thermophilic	60	260–268[b]	152-156*	52.3	Kang et al. (2022)
PLA	FW	Thermophilic	35	21	14**	N.A.	Yu et al. (2023)
PLA	FW	Thermophilic	60	183–228[b]	714-892*	13.7	Kang et al. (2022)
Polyethylene	FW	Thermophilic	35	16.1	14**	N.A.	Yu et al. (2023)

[a] 50% PLA, 50% starch-based shopping bags.
[b] $mLCH_4$ $gCOD^{-1}$.

On the other hand, plastics such as PE and starch-based bioplastics did not enhance the anaerobic digestion process (Table 5). Although most studies have been carried out at laboratory scale and under batch conditions with relatively high degradation efficiencies (>55%), continuous systems at laboratory scale and full scale showed lower degradation efficiencies (<50%). For instance, Kosheleva et al. (2023) studied the mesophilic anaerobic co-digestion of synthetic food waste and cellulose acetate-based bioplastic under semi-continuous conditions in a 3 L digester, obtaining a CH_4 yield of 331 $NmLCH_4$ gVS^{-1}, which was similar to that obtained from the anaerobic digestion of food waste alone (326 $NmLCH_4$ gVS^{-1}) (Kosheleva et al., 2023). It is also important to note that the tested bioplastic exhibited a weight loss of ~45% after 80 days of mesophilic anaerobic digestion, suggesting that cellulose-acetate pretreatment was required to improve its biodegradability. Nonetheless, this value was lower than its batch counterpart, where 98% of the same bioplastic was degraded. Similarly, Gadaleta et al. (2023) reported the mesophilic anaerobic digestion of CA bioplastic at full scale, where a disintegration between 36 and 50% of CA was observed (Gadaleta et al., 2023). This disintegration was lower than that observed in batch assays, where values between 54 and 73% were recorded (Gadaleta et al., 2022). On the other hand, Benn and Zitomer (2018) studied the mesophilic anaerobic co-digestion of PHBs and synthetic municipal primary sludge under continuous conditions and reported that the addition of untreated PHBs and thermochemically pretreated PHBs (≥55 °C, pH ≥ 10, ≥24 h) increased CH_4 production by 5% and 17%, respectively (Benn and Zitomer, 2018). Moreover, the co-digestion stimulated the bioconversion of bioplastics to CH_4 with efficiencies of 80–98%. Finally, it is important to highlight that not all bioplastics are susceptible to biodegradation by anaerobic digestion, and even the results differ at different scales. In this regard, the application of pretreatments accelerates the hydrolysis of bioplastics, which ultimately boost CH_4 production rate. For example, it has been shown the potential of alkaline pretreatment to shorten the anaerobic biodegradation of PHAs to less than one week (*versus* 31–50 days without pretreatment) and to make PLA anaerobically biodegradable under mesophilic conditions (Garcia-Depraect et al., 2023). More recently, Im et al. (2024) investigated the mesophilic anaerobic digestion of various bioplastic materials subjected to hydrothermal pretreatment (150 °C for 3 h). The authors reported methane yields of up to 460, 545, 175, and 490 NmL CH_4 g VS^{-1} for PLA, PHA, PBAT, and PBS, respectively, which were significantly higher than those of untreated bioplastics (Im et al., 2024).

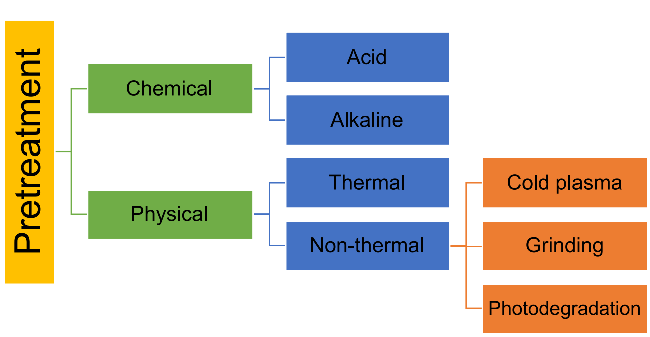


Fig. 8. Classification of bioplastic pretreatments.

4.1.2. *Pretreatments of bioplastics*

Bioplastic pretreatments increase the surface area and reduce the degree of crystallinity and molecular mass of the bioplastic, and can even solubilize the constituents of bioplastics (García-Depraect et al., 2021). Different pretreatments have been recently reviewed by (Mat Yasin et al., 2022; García-Depraect et al., 2021), which can be classified as shown in Fig. 8.

Among all pretreatments tested with bioplastics, alkaline pretreatment stands as a promising technique to improve the biodegradation of bioplastics via hydrolysis (Garcia-Depraect et al., 2023). Hydrolysis is considered as the rate-limiting step in the anaerobic digestion of bioplastics, while the hydrolysis efficiency of alkaline pretreatments has been reported to be, in average, of 70% and 90% for PHA and PLA, respectively (Mat Yasin et al., 2022). Nonetheless, alkaline pretreatment requires long exposure times of 4 h (Yu et al., 2005) to 25 days (Garcia-Depraect et al., 2023), depending on the nature of the bioplastic, the alkaline concentration and the particle size. The long exposure time can be reduced by increasing the concentration of the alkaline chemical or by simply reducing the particle size of the bioplastic. For instance, the solubilization of PHB was less than 5% when exposed to 0.1 N NaOH for 4 h, but when the NaOH concentration was increased to 4 N, PHB solubilization increased to over 70% (Yu et al., 2005). PLA reached 97–99% of solubilization when exposed to 10 M NaOH for 15 days. At this point, it is important to highlight that increasing concentrations of bioplastics require increasing concentrations of alkali to maintain high solubilizations efficiencies. Indeed, the solubilization degree of 100, 150 and 200 g L^{-1} of PLA, PLA/PCL blend, PHB and poly (3-hydroxybutyrate-co-3-hydroxyvalerate) (PHBV) increased significantly when the NaOH concentration was increased from 1 M to 3 M (Garcia-Depraect et al., 2023). In this context, the degree of solubilization that a plastic achieve during pretreatment is a key factor for anaerobic digestion, especially to shorten the lag phase. Recently, Garcia-Depraect et al. (2023) reported that alkaline pretreatment of PLA and PLA/PCL, PHB, PHBV and PHBH significantly enhanced their biomethanization rate compared to their untreated counterparts, estimating that the conversion of PLA and PHBs to CH_4 required between 6 and 25 days to achieve efficiencies over 80% (Garcia-Depraect et al., 2023). Additionally, the particle size plays an important role during bioplastics pretreatment and anaerobic digestion and higher methane yield. Ashraf Joolaei et al. (2024) recently demonstrated that when the particle size of PLA was reduced up to 0.5 mm (Ashraf Joolaei et al., 2024) the methane yield was 320 mL_{CH4} g COD.

In this regard, the application of appropriate alkaline pretreatments to bioplastics could reduce the time needed for anaerobic biodegradation to HRT ≤30 days, similar to those typically implemented in real full-scale anaerobic digesters treating OFMSW, livestock manure or sewage sludge with high efficiencies. However, the scalability of these techniques has not been proven yet and the reduction in particle size will require additional operating cost in real anaerobic digestion plants. Thus, further studies on the pretreatment of bioplastics are needed to investigate the potential of combining thermal-alkaline pretreatment, reducing bioplastics particle size, as well as the co-digestion of organic waste along with alkaline pretreated bioplastics, and how the high alkali concentrations used during bioplastic pretreatment would impact the performance of the co-digestion process before taking pretreatments to industrial levels.

Regarding life cycle assessment (LCA) investigating the environmental impacts of bioplastics management via anaerobic digestion as an end-of-life platform, Hobbs et al. (2021) compared the LCA of the anaerobic co-digestion of 1) PLA and food waste and; 2) pretreated PLA and food waste. For scenario 1) it was considered that PLA was not completely degraded and therefore the digestate was contaminated and sent to landfill. On the other hand, pretreated PLA in scenario 2) met the standards requirements and was used as soil amendment. With the aforesaid, even if both scenarios contributed to eutrophication due to phosphate emissions to the groundwater, scenario 2 had a net impact due to the fertilizer production. Additionally, lower ecotoxicity, global warming potential, human health and cumulative energy demand were also attributed to scenario 2, confirming the beneficial impact of pretreatment prior bioplastics anaerobic digestion (Hobbs et al., 2021). However, PLA production was not considered in the study, which could give an unrealistic approach to the environmental impact of PLA. In this sense, Durkin et al. (2019) compared the techno-economic viability and environmental sustainability of polystyrene and poly(limonene carbonate) (PLC) produced from citrus waste. The results from the model concluded that PLC offers a more sustainable practice, especially in the categories of climate change and fossil depletion. However, PLC entailed a negative impact in the production stage, since the culture of citrus trees requires a considerable water consumption and use of fertilizers, which can lead to eutrophication (Durkin et al., 2019). Certainly, special efforts are needed on finding more sustainable production practices that can make bioplastics ecological favorable materials. In this holistic approach, Rostkowski et al. (2012) proposed a system where PHB was first synthetized from methane and then biodegraded via anaerobic digestion to produce biogas. This approach resulted in lower energy required to produce PHB (37.4 MJ kg^{-1}) compared to PHB produced from corn (41.9 MJ kg^{-1}). This energy saving could result in a global warming potential of −1.94 $kgCO_2$ eq and up to −6.0 $kgCO_2$ eq. However, the main challenge relies on the recovery of PHB and demand of energy (Rostkowski et al., 2012).

4.2. Composting

Bioplastics stand as an eco-friendly option to replace fossil-based plastics. However, their final disposal remains a controversial issue. Even if some bioplastics are certified as compostable, the reality is that they require longer residence times to be fully degraded. Very few studies have been reported on the aerobic degradation of bioplastics in full-scale composting plants, and the results obtained are contradictory. For instance, Lavagnolo et al. (2020) studied the degradation of starch-based bioplastic bags (30% starch, 70% PBAT) via composting in a small-scale pyramidal composting facility. After 55 days, the biodegradability of the bioplastics was lower than the requirements for composting (≤10%, size >2 mm) (Lavagnolo et al., 2020). Similar findings were reported by Gadaleta et al. (2023), who investigated the biodegradation of CA and a composite of CA and (CA-LDH) by full-scale composting (4–5 weeks, 60 °C). At the end of the composting period, neither bioplastic showed more than 20% degradation. Interestingly, the authors also evaluated the biodegradation of CA and CA-LDH when combined anaerobic digestion and composting and found that after both treatments CA and CA-LDH reached 58 and 40% degradation, respectively (Gadaleta et al., 2023). Similarly, Cucina et al. (2022) investigated the biodegradability of starch-based shoppers using thermophilic anaerobic digestion followed by composting and composting alone. The authors reported higher degradation with an integrated treatment consisting of anaerobic digestion followed by composting. The authors reported that the starch-based shoppers exposed to thermophilic anaerobic digestion for 30 days followed by 60 days of composting were completely degraded, while the starch-based shoppers composted directly only were 98% degraded in the same period (Cucina et al., 2022). Contrary results were reported by Bandini et al. (2022b), who evaluated the pilot-scale anaerobic digestion followed by composting of OFMSW with PLA and SBB. After thermophilic anaerobic digestion and composting of the digestates for 22 days at 65 °C, the degradation of the bioplastics did not comply the UNE EN 13432 standard (Bandini et al., 2022b). More recently, Falzarano et al. (2024) investigated the composting of digestates separately generated from thermophilic anaerobic digestion of PLA and Mater-Bi® bioplastics. After 40 days of anaerobic treatment, the biodegradability for PLA and Mater-Bi® bioplastics was 92% and 45%, respectively. However, the combination of anaerobic digestion and composting provided additional stabilization of the organic material and resulted in almost complete mineralization of the remaining bioplastic fraction (Falzarano et al., 2024). In this context, full-scale biodegradation of bioplastics is likely to require an initial anaerobic digestion step coupled with composting as a secondary step, as the highest efficiencies of plastic degradation have been reported during the anaerobic digestion process. The combination of anaerobic digestion and composting results in longer residence times, which might favor the ultimate degradation of bioplastics.

Also, the type of co-substrate in both the anaerobic digestion and composting stages should be evaluated in further studies to ensure compatibility between mixtures of different bioplastics and organic wastes (Falzarano et al., 2024; García-Depraect et al., 2024). In this context, co-substrates may help the biodegradation of bioplastics under composting conditions by maintaining higher moisture in the system, enhancing microbial activity due to the presence of easily biodegradable substances, and/or promoting the chemical hydrolysis of bioplastics (Falzarano et al., 2024).

5. Anaerobic capacity available in Europe and countries around the world

To date, the reported studies on the anaerobic biodegradability of bioplastics have shown promising results especially when bioplastics are pre-treated before undergoing the biological degradation or when co-digested with different substrates. However, one of the major challenges is the scalability of the experimental procedures to real life. Thereby, methods with realistic techno-economical approaches and the currently available infrastructure must be considered to draw a roadmap towards bioplastics' treatment. In this sense, the increasing government policies and International Agreements that incentivize biogas and biomethane production have positively impacted this energy sector. Thereby, the increased installed capacity in the EU-27 and some

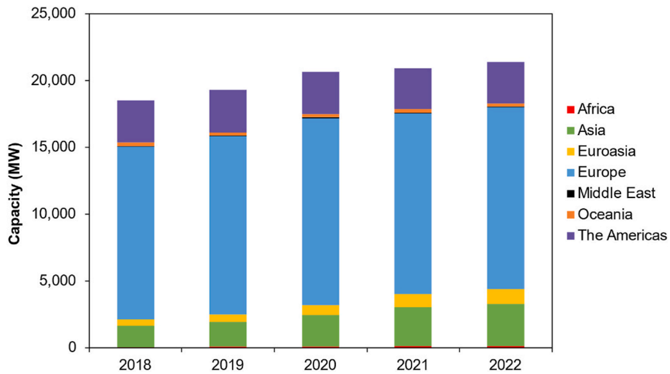


Fig. 9. Worldwide biogas installed capacity, (IRENA, 2021).

countries around the world positions anaerobic digestion as the most feasible treatment for bioplastics. This section provides an overview of the current and upcoming trends regarding biogas and biomethane installed capacity. Special attention is devoted to the types of substrates treated in most of the existing plants.

The global installed biogas production capacity has increased in recent years, with Europe being the leading region, accounting for more than 60% of the global capacity (Fig. 9) (IRENA, 2021). This continuous growth is mainly driven by the policies and incentives in the European Union Member States. Indeed, the EU-27 accounts for more than 80% of the total installed capacity in Europe (IRENA, 2021; Scarlat et al., 2018). In fact, the latest statistical report from the European Biogas Association (EBA) shows that about 21 billion cubic meters (bcm) of biogas and biomethane (from anaerobic digestion) were produced in Europe in 2022, an amount equivalent to the total natural gas consumption of Belgium.This figure is expected to increase up to 35 bcm by 2030 (EBA, 2023).

In 2023, the EBA reported the existence of more than 20,000 biogas and biomethane plants in Europe, of which 1323 are estimated to be biomethane plants. Since 2011, biomethane plants have experienced a rapid growth. From 2020 to 2021, biomethane production increased by 20% and is expected to continue to grow faster than biogas plants. Considering that biomethane plants can produce 4 times more energy (35 GWh) per year compared to biogas plants (8 GWh), biogas plants are being converted into biomethane plants. It is important to highlight that biogas plants refer to the production of biogas via anaerobic digestion, whilst biomethane plants refer to the coupling of biogas production (via anaerobic digestion) and biogas upgrading to produce biomethane with CH_4 contents >95% for injection into the natural gas grid or use as vehicle fuel. In 2022, 22% of upgraded biogas was used in buildings. A further 14% was used in industry, 19% for transport and 15% for power generation Moreover, the similarity of biomethane to natural gas has also influenced its production, as 58% of the produced biomethane is injected into the natural gas grid and 19% into the transport grid (EBA, 2023).

5.1. Installed biogas capacity and number of plants

The installed biogas capacity of the most representative countries in Europe, Asia, Latin America and Oceania is shown in Fig. 10a. Germany has positioned itself as the leading country in Europe and worldwide in terms of biogas (71 TWh in 2021) and biomethane (12.8 TWh in 2021) production, followed by Italy and France (Fig. 10b) (EBA, 2022). However, special attention should be paid to Denmark and Switzerland, which generated biogas to fulfill 24% and 15% of their total gas consumption in 2021, respectively, ranking first among European countries. In addition, Denmark, Sweden, Norway and Estonia have more biomethane plants than biogas plants in Europe (EBA, 2022).

In 2021, Germany reported over 11,000 biogas plants and by 2023 reported 254 biomethane plants (EBA, 2023, 2022). Since 2021, the German government has abandoned the Feed-in-Tariff (FiT) and the German biogas production has started to decline, but small units with maximum capacity of 100 kW are still eligible for FiT supports. Although the German biomethane industry is growing slower than in Denmark, France, the Netherlands and Italy, Germany is expected to become one of the main producers of liquefied biomethane (Bio-LNG) (EBA, 2023). In 2021, the Netherlands was home of 260 biogas plants and 70 biomethane plants (EBA, 2023). In 2021, up to 20% of the Dutch biomethane produced was used in the transport sector and is expected to grow in the coming years as the country invests in the production of Bio-LNG, which is expected to boost biogas production.

Italy is the second country in the EU-27 in terms of number of plants and production of biogas. In 2021, Italy accounted 1800 biogas plants

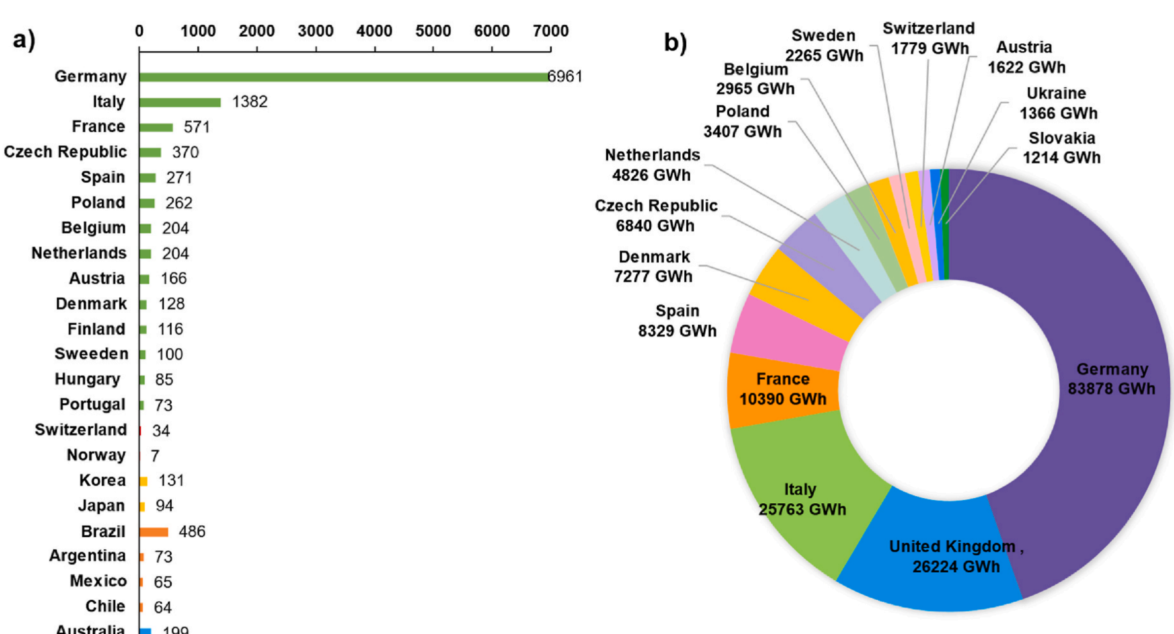


Fig. 10. a) Installed biogas capacity of representative countries around the world; b) Major biogas producers in Europe (IRENA, 2021).

and 33 biomethane plants (EBA, 2023, 2022). Italy is expected to be among the top 3 and top 5 EU-27 countries in terms of biomethane production by 2030 and 2050, respectively (Alberici et al., 2022). The majority of Italian biomethane is currently used in the transport sector and this trend is expected to continue. Some projections estimated that Italy will become one of the leading countries of Bio-LNG producers in Europe. In 2022, Norway was home of 17 biogas plants and 3 biomethane plants (EBA, 2023, 2022). In recent years, the biomethane sector has grown mainly due to its use in the transport sector in the form of Bio-LNG. Indeed, Norway has pioneered the Bio-LNG in Europe and is expected to become a leading producer in the coming years.

The biogas and biomethane sector in France is growing at a high rate. Indeed, France operated 945 biogas plants by the end of 2021, while by April 2023 France reported the highest number of biomethane plants (477) in Europe (EBA, 2023). The Energy Transition for Green Growth law and the Long Term Energy Schedule in France, set targets to increase the share of renewable gases to 10% and an injection target of 7–10% of biomethane by 2030. In this context, France is expected to have the highest biogas and biomethane potential in Europe by 2050 (Alberici et al., 2022). Denmark is one of the fastest-growing producers of biomethane, with biomethane production exceeding biogas production in 2018. Nowadays, Denmark has 51 biomethane plants in operation and it is expected that by 2030 biomethane will supply 100% of the Danish gas demand. On the other hand, the Czech Republic is another growing nation in terms of biogas and biomethane production. In January 2020, the Czech government stablished the Act on Promoting Renewable Energy Sources, which supports biogas and biomethane production. In 2021, the Czech Republic reported 573 operational biogas plants and 2 biomethane plants. Recently, most of the Czech biogas produced is used in combined heat and power (CHP) systems, and it is expected that the biomethane produced will be used in the transportation sector. Poland is another country that plans to expand its biomethane sector in the coming years, with 346 biogas plants in operation by the end of 2021. In Poland, biomethane is expected to be used mainly in the transport sector. On the other hand, Belgium registered 8 active biomethane plants and 189 biogas plants by 2022. The interest in producing biomethane started to increase in 2018, and even if there are no specific targets for biomethane production, it is estimated that Belgium can achieve 17% of its production potential by 2030. By the end of 2021, Austria registered 423 biogas plants and 16 biomethane plants. Recently, Austria has been shifting from biogas to biomethane production with the aim of injecting biomethane into the Austrian gas grid. In this way, Austrian biomethane plants and biomethane production are expected to grow in the next years. In Finland, the number of biogas plants has increased since 2011, and by 2021, Finland was home of 108 biogas plants and 23 biomethane plants. Up to 40% of the Finnish biogas produced is used in CHP systems and in heating applications. Concomitantly, the Finnish biomethane sector has grown significantly and is expected to continue growing, mainly due to the support of the national biogas action plan. By the end of 2021, Sweden hosted 207 biogas plants and 72 biomethane plants. Since mid 2022, Sweden has had a long-term production support for biomethane which has increased the number of biomethane plants. Currently, Swedish biomethane is mainly used in the transport sector. Finally, Swiss biogas and biomethane production has increased since 2011. Thus, Switzerland reported 418 biogas plants and 40 biomethane plants by the end of 2021. In 2021, 27% of the total gases used in the Swiss transportation was covered by biomethane.

Spain has the 4th highest biogas and biomethane potential in the EU-27. By the end of 2021, Spain hosted 250 biogas and 4 biomethane plants and by 2022, the number of biomethane plants increased to 5. However, Spain potential has not yet been fully exploited. To date, the biogas sector in Spain has been mainly driven by the environmental needs (*i.e.* waste treatment) and/or private consumption. The next years will be important for the development of the biomethane sector in Spain, considering that the biogas route agenda published in 2022 will be upgraded to fulfill the requirements of the REPowerEU plan. Since 2014, Portugal has been supporting the production of biogas and biomethane, and its biogas production has been gradually increasing. By the end of 2021, there were 63 biogas plants in Portugal, although Portugal currently has no biomethane plant, mainly because there is no specific support for Portuguese biomethane production. Nevertheless, a national strategy to promote biomethane production in Portugal is expected to be published in the next few years. Finally, Hungary has an underdeveloped biogas industry, with only 40 biogas plants in operation. The 2030 Hungarian targets stablish that at least 3.5% of the total energy consumption for the transport sector must come from second-generation biofuels and biogas (IEA, 2022), suggesting that the number of plants in Hungary could increase in the coming years. According to the IEA, Hungary has a large potential for biogas production, but incentives are needed to develop this sector.

The Japanese biogas market is in its infancy, accounting for 1.5% of biomass energy (Bourgogne, 2021). Today, Japan has 380 biogas plants and 6 biogas upgrading plants. Additionally, small agricultural digesters have been also installed in municipalities and cooperatives, but this figure is not well documented. Unlike most European countries, Japanese biogas is mainly used for electricity generation. One of the major drawbacks facing the biogas market in Japan is that most of biogas plants are small, averaging 350 kW, and most waste is incinerated (Bourgogne, 2021). Similarly, in South Korea, biogas and biomethane represent a small share of the total power generation. It is estimated that there are 132 biogas plants in South Korea, where government support is critical to the development of this green energy platform technology (International Energy Agency, 2020).

In Latin America, Brazil operated 811 biogas plants in 2021 (CIBiogas, 2023). Brazil has a huge potential for biomethane production due to its large biomass production potential. Indeed, it is estimated that Brazil can produce up to 121 million of m^3 d^{-1}, but currently only less than 1% is produced (ABiogás, 2016). On the other hand, Argentina has increased its installed biogas capacity in the last decade (IRENA, 2021). In April 2020, the Argentine Chamber of Renewable Energies (CADER) proposed a national law to promote the production and injection of biomethane to the natural gas grid, with the goal of achieving a 5% share of biomethane in total gas consumption by 2030 (CADER, 2020). Today, Argentina has 27 biogas plants in operation. However, the Argentine government is currently investing in anaerobic digestion technology, and therefore biogas and biomethane production could increase significantly in the coming years. Chile is another Latin American country that is investing in biogas technology. The installed biogas capacity in Chile has increased over the last decade and it is estimated that Chile now has 39 biogas plants. However, according to the Chilean agricultural network, anaerobic digestion is being implemented as a waste treatment technology rather than an energy production technology. Finally, biogas and biomethane production in Mexico has not increased significantly in recent years. It is estimated that there are currently 345 biodigesters in operation in Mexico, but only for internal waste management and energy production (Ramírez-Higareda et al., 2019).

In Australia, the biogas industry is beginning to grow. In 2017, Australia was home of 242 biogas plants, and in 2020, Australia introduced the first biomethane plant at the Malabar wastewater treatment plant. This plant was announced to inject its biomethane into the gas distribution network in 2023. In 2021, Australian energy ministers agreed to include blends of hydrogen, biomethane and other renewable methane gas mixtures in the national energy regulatory framework.

5.2. *Type of substrate*

The substrate used for biogas and biomethane production varies from country to country, depending on weather, residues, industries, etc. Substrates are typically grouped into six categories: agricultural, sewage sludge, landfill, organic municipal solid waste, industrial food and beverage waste, and others. However, the available data hardly specify whether the agricultural substrate refers to energy crops,

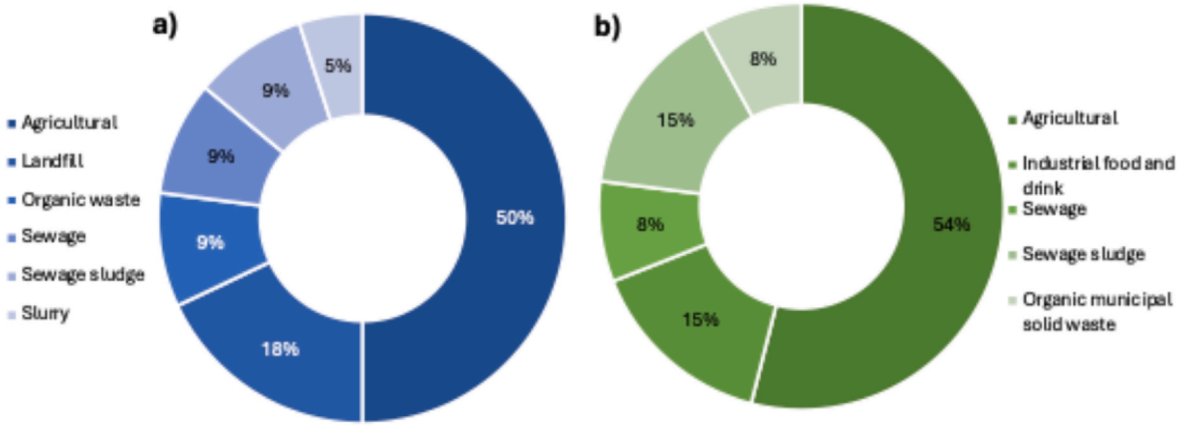


Fig. 11. Main substrates used for a) biogas and b) biomethane production in the countries studied herein (Bourgogne, 2021; CADER, 2020; CIBiogas, 2023; EBA, 2022; Danish Energy Agency, & Institute of Engineering UNAM, 2017).

agricultural residues or manure. Therefore, the specific substrate used for biogas or biomethane production is not defined in this review.

For instance, agricultural residues dominate the biogas market in Argentina, Austria, Belgium, Brazil, the Czech Republic, Denmark, France, Hungary, Italy, Poland, the Netherlands, and Sweden, while energy crops and agricultural residues are the main substrates used in Germany (EBA, 2022). In Australia, Finland, Portugal and Spain biogas is mainly produced from landfilled waste. In Switzerland, Mexico and Norway biogas production is mainly attributed to sewage sludge digestion. Japan and South Korea anaerobically digest food waste and organic waste, respectively. Finally, the main substrate used for biogas production in Chile is slurry and manure. The shares of the main substrates used for the production of biogas (Fig. 11a) and biomethane (Fig. 11b) in the countries herein mentioned are shown above. Most countries have recently made commitments to produce biomethane in a more sustainable way, mainly from agricultural and organic residues, sewage, sewage sludge and slurry.

6. Composting capacity available in Europe and countries around the world

Composting has emerged as a promising technique for the treatment of bioplastics since their incorporation into existing composting facilities does not represent a technological challenge, *i.e.* risks of clogging. However, bioplastics degradation via composting has not yet reached the required standards (EN 13432), since a longer treatment time is required. Moreover, the presence of bioplastics in the final compost represents an environmental and economic risk. In this context, the combined anaerobic/aerobic degradation represents a feasible pathway to treat bioplastics as the joined treatments result in longer biodegradation times. However, according to the ECN, only 5% of the existing composting in Europe were combined anaerobic digestion/composting facilities. Thus, public policies and incentives are necessary to promote their implementation. This section provides an overview of global opportunities related to composting technology. Special attention was given to the composting installed capacity and the main substrates used in the EU-27 and other countries around the world.

6.1. Installed capacity and number of plants

According to the ECN, only 17% (less than 40 million tons) of municipal solid waste (MSW) in Europe is organically recycled into compost and digestate. In 2020, 71 million tons of bio-waste were collected and treated by composting (59%) and anaerobic digestion (41%) in the EU-27, Norway, Switzerland and the United Kingdom (Fig. 12). Green garden and food waste were the dominant feedstocks in composting plants, while food waste and 'other wastes' were mainly used in anaerobic digestion plants (ECN, 2022b).

In its latest report, the ECN estimates that there are 5800 bio-waste treatment facilities in the EU, Switzerland, Norway and the UK, of which 3800 are devoted to composting (European Compost Network, 2022). On average, each composting facility treats 8000 tons of bio-waste per year. In addition, it is estimated that 88% of composting facilities treat only bio-waste, 7% co-compost bio-waste and sewage sludge, and the remaining 5% co-compost bio-waste and anaerobic digestate.

The European Environment Agency (EEA) has classified European countries according to their treatment capacity, as shown in Fig. 13. Additionally, the Organization for Economic Cooperation and Development (OECD) has reported the most common bio-waste treatments of different countries around the world, where the Netherlands has the highest percentage of bio-waste treatment by composting (29%), followed by Italy (26%), Switzerland (23%), Lithuania (23%) and Germany (22%) in 2020 (OECD, 2020).

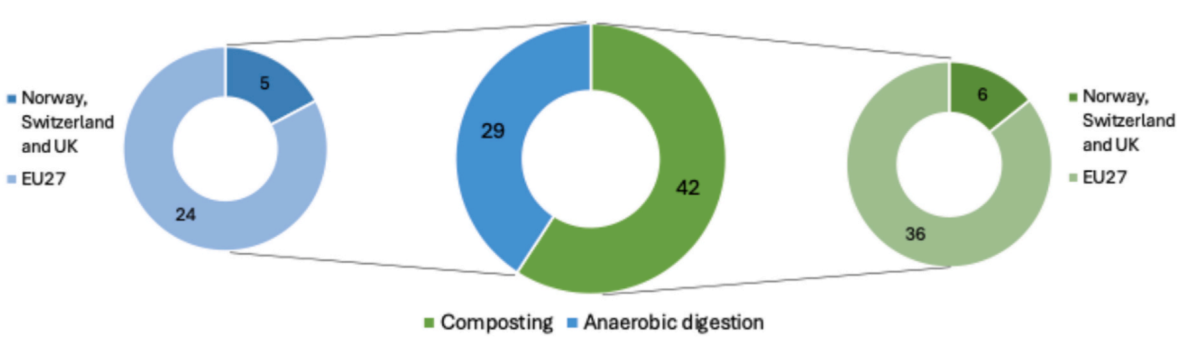


Fig. 12. Bio-waste treated by composting and anaerobic digestion (tons per year) in Europe (European Compost Network, 2022).

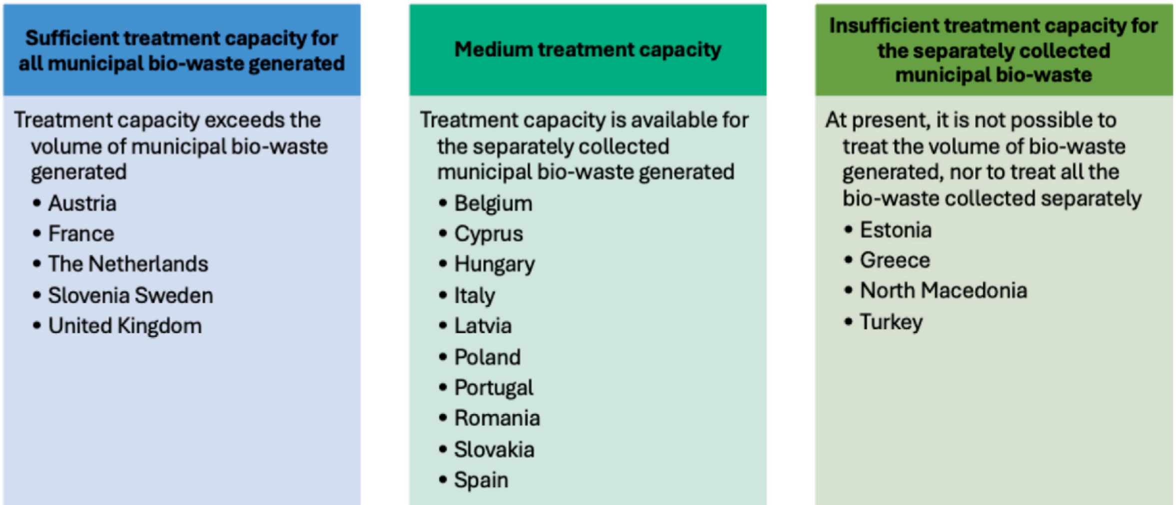


Fig. 13. Classification of European countries according to their bio-waste treatment capacity (van der Linden and Reichel, 2020).

The composting sector in the Netherlands consists of two categories: 1) green waste, which refers to residues from agriculture and green public spaces, landscapes and roadsides, and 2) household bio-waste, which refers to vegetable, fruit and garden waste. In 2018, it was reported that 3.2 million tons of green waste and 1.4 million tons of household bio-waste were collected. Typically, green waste is composted in open windrow composting facilities. It is estimated that there are 100 licensed facilities with treatment capacities ranging from 1000 to 100,000 tons per year. On the other hand, household bio-waste is composted in closed vessel systems and it is estimated that there are currently 21 household bio-waste facilities operating in the Netherlands (ECN, 2018).

Germany started separating household bio-waste and green waste in 1985, and since then the bio-waste recycled has increased steadily. Indeed, up to 15.3 million tons of bio-waste were treated through composting and anaerobic digestion in 2020. It is estimated that in 2020 there were 817 compost plants, 277 digestion plants, 58 combined digestion and composting plants in Germany (ECN, 2023a).

In 2006, national legislation in Italy set a target to separate 65% of municipal solid waste by 2025, and since January 1st' 2022, the Italian national legislation obligates the separate collection of organic waste in Italy. In this way, bio-waste, more specifically food waste from households, is targeted as a key factor to reach the targets set by the national legislation. In 2021, 7.4 million tons of bio-waste and 2 million tons of green waste were collected separately. Furthermore, it was estimated that in 2021 there were 293 composting plants and 63 anaerobic digestion coupled with composting plants in Italy, of which the 10 largest facilities had a capacity to treat more than 100,000 tons per year, representing 25% of the total organic waste treated in Italy (ECN, 2023b).

The three regions of Belgium (Flanders, Brussels and Wallonia) have different waste legislation. For instance, Flanders has paid a lot of attention to bio-waste treatment, composting, anaerobic digestion and sustainable use of compost and digestate. Since the early 1990's, Flanders has started the separate collection of bio-waste and since the late 1990's, the separate collection of household waste from green waste or kitchen and garden waste is mandatory in municipalities. Moreover, it was estimated that in 2021, 44 plants treated 672,000 tons of green compost, 10 plants treated 407,000 tons of kitchen and garden waste, and 38 anaerobic digestion plants treated 1.54 million tons of bio-waste (ECN, 2023c).

In 2020, Austria treated 1.7 million tons of organic waste, by-products and industrial residues in 404 composting facilities with a treatment capacity of up to 1.68 million tons. In addition, Austria has a nationwide bio-waste collection system, and an estimated 1.5 million tons of organic material was composted in home and community facilities (ECN, 2023d).

In Denmark, recycling is the most important waste treatment, followed by incineration with energy recovery and finally landfilling. It is estimated that the total amount of waste in Denmark increased from 11.5 to 12.5 million tons between 2016 and 2018. In particular, food waste from households increased from 61,700 in 2017 to 95,300 in 2018 (ECN, 2020a). By 2020, Denmark treated 19% of its municipal waste through composting (OECD, 2020), and although anaerobic digestion has a long tradition in Denmark, the need to recycle nutrients such as phosphorous makes composting technology more attractive for bio-waste treatment (ECN, 2020a).

In 2021, Poland treated 13% of its municipal waste by composting. Even if the Polish composting capacity is available for the collected bio-waste, the capacity for anaerobic digestion is higher than for composting (OECD, 2020; van der Linden and Reichel, 2020). On the other hand, Portugal has a similar capacity for composting and anaerobic digestion, although Portugal has combined facilities, where bio-waste is first digested and then the digestate is composted (van der Linden and Reichel, 2020).

In Norway, the source separation of bio-waste started in the 1990's and since then the amount of bio-waste collected has increased. For instance, in 2011, 171,000 tons of household bio-waste was collected and in 2016 this number increased to 333,000 tons. In 2016, 22,8000 tons were treated by composting and 105,000 tons by anaerobic digestion. In 2017, Norway had 40 composting plants for the treatment of food waste, sludge or garden waste (ECN, 2017a).

By 2018, it was estimated that 50% of Swedish food waste would be separated, collected and treated by anaerobic digestion (40%) and composting (10%). In 2015, Sweden reported 40 composting plants, 11 of which treated food waste. Despite this, food waste has changed this trend and is preferably treated by anaerobic digestion (ECN, 2017b).

In Finland, bio-waste treatment has been focused on composting, but anaerobic digestion has become the preferred option in recent years. Finnish bio-waste consists mainly of household kitchen waste, commercial bio-waste and catering waste. The Finnish government encourages home composting for individual households and garden waste. In 2017, 391,000 tons of municipal bio-waste were collected, of which up to 239,000 tons were composted. In 2017, Finland reported 20 in-vessel composting facilities for bio-waste treatment with a capacity of 5000 to 50,000 tons per year. Additionally, 160 composting facilities were devoted to the composting of sewage sludge (ECN, 2019).

France has sufficient capacity to treat all of its generated bio-waste.

In 2021, France treated 19% of its municipal waste by composting (OECD, 2020; van der Linden and Reichel, 2020). In 2021, Czech Republic treated 12% of its total municipal waste by composting. Czech Republic has the second largest production of garden waste in its bio-waste from Europe, whilst its generated food waste is half of the EU average (OECD, 2020; van der Linden and Reichel, 2020). Spain treated 17% of its total organic waste by composting. Spain is able to treat all of its separated bio-waste, however if the collected bio-waste increases, its treatment capacity must increase accordingly. Nonetheless, Spain has recently built the largest composting plant in Europe in Murcia, which has the capacity to treat 140,000 tons per year of sludge (van der Linden and Reichel, 2020). In 2021, Switzerland composted 22% of its total municipal waste, and up to 90% of the Swish compost was used in agriculture (OECD, 2020; van der Linden and Reichel, 2020). Hungary exhibits one of the lowest production of bio-waste per person (75 kg), and this bio-waste is mainly composed of garden waste, food waste and other bio-wastes. Hungary is capable to treat all of the separated bio-waste, but if the separated bio-waste increases the infrastructure must be extended accordingly.

Korea has one of the most advanced waste management systems in the world, and according to the Ministry of Environment 86% of Korean waste is recycled. Moreover, 95% of the generated food waste is currently recycled. South Korea has a mandatory composting system which has resulted in many home composting systems. In Japan the use of composting began after World War II, but incineration became the dominant waste treatment method in the 1970s. However, the number of composting plants increased in the 1980s, with livestock manure, sewage sludge and agricultural waste as the main compostable wastes. By 2020, Japan had 100 composting facilities in operation (Kawai et al., 2020).

In 2021–22, Australia produced 14.8 million tons of organic waste and up to 7.7 million tons were treated through composting (AORA, 2021). According to the Australian Organics Recycling Association, Australia has 305 operating organics recycling facilities, and home composting is also used.

In Argentina, 50% of the municipal waste is organic fraction. Argentina has composting plants in different locations of the country and in most of them, the process is carried out with minimal equipment. The biggest composting plant (with a treatment capacity of 800-1100 tons per month) is operated by a private company (Holland Circular Hotspot, 2021). On the other hand, the composting sector in Brazil has a great potential. Most of the Brazilian composting plants use windrow systems, although aerated static piles are also used, especially to prevent odors when composting sewage sludge and food waste. Brazilian composting facilities have a capacity to treat 90,000–170,000 tons of waste per year (De Schueler and Mahler, 2003). In Chile, up to 58% of the household waste is organic. In 2021, Chile's National Organic Waste Strategy established that 66% of its organic waste should be recovered through composting by 2040. To achieve this target, Chile has partnered with Canada to receive economic and educational support (CCAC, 2021). Finally, Mexico generated 44 million tons of waste in 2017, with an estimated organic fraction of 46%. Mexico has 19 composting plants capable of treating 27 tons of bio-waste per day (Holland circular hotspot HCH, 2021).

6.2. Type of substrate

The type of substrate treated by composting depends strongly on the country and its legislation. For instance, bio-waste in Hungary consists mainly of garden waste, while bio-waste in Denmark is mainly composed of food waste (van der Linden and Reichel, 2020). However, the main substrates used in composting in general are green waste, garden waste and food waste from households (Fig. 14a) (European Compost Network, 2022).

Food waste is the main substrate produced in Denmark, Finland, Germany, Italy, Korea, the Netherlands, Norway, Poland, Portugal, Spain and Sweden. However, most of this food waste in these countries is treated by anaerobic digestion (van der Linden and Reichel, 2020). On the other hand, Australia, Belgium, the Czech Republic, France, Hungary and Switzerland mainly produce garden waste, most of which is treated by composting (van der Linden and Reichel, 2020). In Japan, composting facilities mainly treat sewage sludge, livestock manure and agricultural waste (Kawai et al., 2020). In Austria, household bio-waste is the main substrate collected and composted (ECN, 2020b). Countries such as Argentina, Chile, Brazil and Mexico defined bio-waste as "organic waste", which mainly consists of food waste and garden waste.

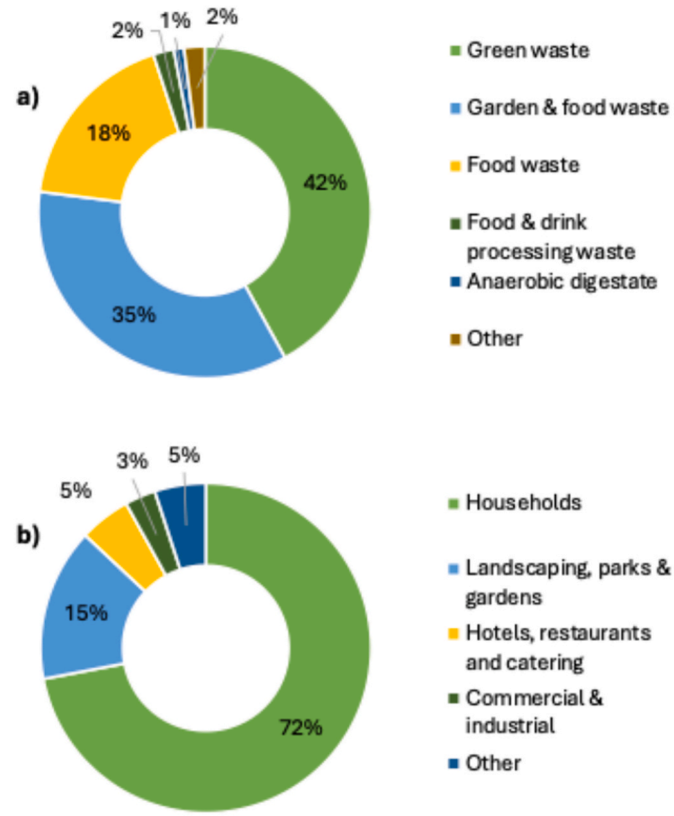


Fig. 14. a) Types of bio-waste collected for composting treatment, b) Sources of bio-waste treated in composting plants (European Compost Network, 2022).

Many countries have made efforts to separate the OFMSW in order to facilitate its disposal and improve the quality of compost and digestate. In composting, green, garden and food waste are typically the dominant substrates, but many recent national policies aim to reduce food waste, which means that green waste would become the dominant substrate.

Finally, the main sources of bio-waste in the target countries in this review are households, followed by landscaping, parks and gardens (Fig. 14b). Some countries, such as South Korea, are forcing their populations to adopt home composting, with the aim of reducing waste generation and further treatment in centralized facilities. Although this is a utopian idea, it requires the participation and education of the entire population and will take many years to reduce household waste disposal.

7. Challenges and future perspectives

In 2018, with the publication of the European Strategy for Plastics in a Circular Economy, the European Commission took a step towards a more resource-efficient system to address the challenges of plastics, from their production to their final disposal. This strategy aimed to reuse and recycle plastics, with a particular focus on the collection and treatment, in order to significantly reduce the presence of plastics in the environment. This ambitious strategy aimed at increasing the recycling capacity

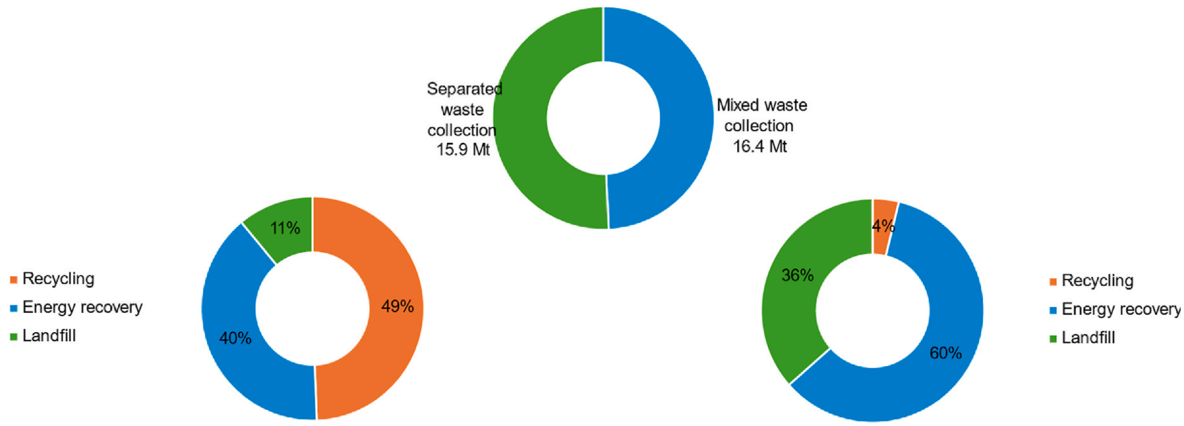


Fig. 15. Plastics waste collection and treatment. Note: energy recovery refers to incineration (Plastics Europe, 2024).

of plastics in the EU, and it is expected that by 2030 the recycling capacity of plastics will be 4 times higher than in 2015. In this context, the production of bio-based and biodegradable plastics remains one of the main priorities to achieve a circular economy for plastics.

Particular efforts have been implemented in bioplastics intended for packaging since this sector represents up to 40% of the bioplastics production share. In this regard, the EU Waste Framework Directive 2008/98 EC and the Packaging and Packaging Waste Directive (94/62/EC) has obligated all member states to recycle and prepare for re-use up to 70% of municipal waste (bioplastics included) by 2030 and to phase out landfill of recyclable waste (i.e. bioplastics). However, the misinformation and misunderstanding about the difference between bio-based and biodegradable plastics has not significantly reduced the problem originally caused by their fossil-based counterparts. Considering that not all bioplastics are biodegradable and not all biodegradable plastics are bio-based, the sole bio-based nature does not meet the expectations of the circular economy of bioplastics (García-Depraect et al., 2021).

Additionally, their inappropriate collection and separation, has limited bioplastics recycling processes to mechanical recycling and incineration (Fig. 15). Nonetheless, when bioplastics are properly separated the share of recycling has a 10-time increment compared to the bioplastics waste that is not separated (Plastics Europe, 2024). The proper separation and collection of bioplastics is fundamental to achieve their circularity. In this way, the participation of citizens and well-stablished public policies that drive the proper disposal of bioplastics is of paramount relevance. Since de-plasticization is something unrealistic and even impossible today, a good communication between science-industry-policy-citizens about the nature of bioplastics would facilitate their correct disposal as well as their reuse, which would significantly minimize the presence of bioplastics in the environment.

Although the bioplastics treatment is mainly directed to mechanical processes and incineration, it is important to address that these two technologies do not necessary entail sustainability. For instance, a life cycle assessment (LCA) conducted comparing incineration, landfill and mechanical recycling demonstrated that incineration has the highest negative environmental impact, while recycling entails an environmental benefit (Hou et al., 2018). Moreover, another LCA comparing mechanical treatment and anaerobic biodegradation with CH_4 recovery demonstrated that mechanical treatment has a negative impact on human health and on the environment as a result of the washing process, which is highly energy and water demanding. On the other hand, anaerobic biodegradation with CH_4 recovery for energy generation entails negligible impacts on the environment (Martín-Lara et al., 2022).

However, despite the promising potential of anaerobic digestion, the bioplastics present an incomplete biodegradation during the typical retention times of anaerobic digesters, which has increased the concern and rejection of bioplastics by the owners of anaerobic digestion plants. In this sense, research has been conducted in the coupling of other waste treatment technologies such as gasification or pyrolysis, to create a synergistic effect that aims at increasing the waste treatment efficiency. Several studies have been reported suggesting that the combination of anaerobic digestion and pyrolysis can improve the sustainability of digestate management, especially if biochar is produced (Tayibi et al., 2021). However, even if anaerobic digestion is a well-stablished technology, pyrolysis is still in its early stages and bioplastics waste treatment needs realistic solutions. In this sense, anaerobic digestion and composting symbiosis stand as a more feasible pathway for bioplastics biodegradation. Additionally, the increasing installed aerobic and anaerobic capacity as well as the increasing number of combined anaerobic digestion and composting facilities position biological treatments as a reliable end-of-life for food contaminated bioplastics. Nonetheless, government subsidies and incentives should be mandatory to foster biological treatment technologies, especially in countries where landfilling is the most common practices for food-contaminated bioplastic waste. It is important to highlight that biological treatments techniques should be considered only for food-contaminated bioplastics since their reuse or recycling to produce other polymers is difficult. This is because some food contaminants are difficult to remove and the washing stage during bioplastics recycling represents a critical stage.

In any case, since to approach a problem output it is necessary to focus on the input, in the bioplastic industry context, it is necessary to look for more environmentally compatible materials that are easy to manage. Albeit bioplastics such as PLA, PHAs, or starch-based are considered biodegradable, the reality is that their biodegradability rate is low, limiting their biological treatment. Studies have reported some discrepancies when scaled up in real aerobic and anaerobic treatment plants. In fact, recent studies have demonstrated that the complete degradation of bioplastics is not achieved within the typical treatment times, and biodegradations ranging from 50 to 70% have been achieved in retention times between 60 and 112 days, which could be a risk in the operation of real treatment plants, especially in real anaerobic digesters. Although pretreatments strategies (i.e. alkali pretreatments, grinding, etc.) are regarded as a solution to improve bioplastics biodegradation, their implementation under real-scale applications represents an unrealistic challenge.

Therefore, the development of materials that comply with international standards for biodegradation and are compatible with real scale processes represents nowadays the real challenge and goal. In the meantime, the combined anaerobic treatment followed by aerobic treatment is necessary to achieve an effective bioplastics biodegradation.

8. Conclusions

The implementation of the EU Circular Economy plan and the directive (EU) 2018/851 aim to recycle 65% of municipal waste (bioplastics included) by 2035, and to achieve this ambitious target it is necessary to increase a proper separation and collection of biowaste (bioplastics included). To date, 40% of the produced bioplastics are destined to the packaging market however, the misinformation and misunderstanding about the difference between bio-based and biodegradable plastics have limited their proper collection, separation and recycling.

Different recycling techniques are currently available for bioplastics, with mechanical recycling being the prefer recycling technique for separately collected bioplastics of high grade or that are not contaminated with other waste, i.e. food waste. On the other hand, when bioplastics contain considerable amounts of organic food waste, biological treatments i.e. composting or anaerobic digestion, should be considered. Nonetheless, bioplastics degradation require long retention times (up to 100 days) which is incompatible with real anaerobic digestion and composting facilities. Additionally, the presence of residual bioplastics in the digestate and compost represents a risk in their subsequent valorization.

In this context, the coupling of different treatment techniques, such as gasification, pyrolysis and composting to anaerobic digestion process has been proposed to increase the waste treatment efficiency. However, gasification and pyrolysis are in their early stage and their scalability is not realistic yet. Whilst the symbiotic anaerobic digestion-composting system could be a more realistic approach, especially for the increasing number of anaerobic digestion-composting facilities in the EU.

Nonetheless, it is necessary to look for materials that comply with the international standards for biodegradation and are compatible for both home composting conditions and industrialized conditions.

CRediT authorship contribution statement

Laura Vargas-Estrada: Writing – original draft, Investigation, Conceptualization. **Octavio García-Depraect:** Writing – review & editing. **Johannes Zimmer:** Writing – review & editing. **Raúl Muñoz:** Writing – review & editing, Supervision.

Declaration of competing interest

There is no conflict of interest between the authors.

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Data availability

The authors do not have permission to share data.

References

ABiogás, 2016. Biogás e biometano no Brasil.
Abraham, A., Park, H., Choi, O., Sang, B.I., 2021. Anaerobic co-digestion of bioplastics as a sustainable mode of waste management with improved energy production – a review. Bioresour. Technol. 322, 124537. https://doi.org/10.1016/j.biortech.2020.124537.
Ahmad, S., Khalid, R., Abbas, S., Hayat, R., Ahmed, I., 2021. Chapter 6 - potential of compost for sustainable crop production and soil health. In: De Mandal, S., Passari, A.K. (Eds.), Recent Advancement in Microbial Biotechnology. Academic Press, pp. 123–170. https://doi.org/10.1016/B978-0-12-822098-6.00005-7.
Ahsan, W.A., Hussain, A., Lin, C., Nguyen, M.K., 2023. Biodegradation of different types of bioplastics through composting—a recent trend in green recycling. Catalysts 13, 1–14. https://doi.org/10.3390/catal13020294.
Alberici, S., Grimme, W., Toop, G., 2022. A Gas for Climate Report Feasibility of REPowerEU 2030 Targets, Production Potentials in the Member Biomethane Production Potentials in the EU.
AORA, 2021. Vision 2031.
Ashraf Joolaei, A., Makian, M., Prakash, O., Im, S., Kang, S., Kim, D.-H., 2024. Effects of particle size on the pretreatment efficiency and subsequent biogas potential of polylactic acid. Bioresour. Technol. 394, 130306. https://doi.org/10.1016/j.biortech.2024.130306.
Bandini, F., Misci, C., Taskin, E., Cocconcelli, P.S., Puglisi, E., 2020. Biopolymers modulate microbial communities in municipal organic waste digestion. FEMS Microbiol. Ecol. 96, 1–13. https://doi.org/10.1093/femsec/fiaa183.
Bandini, F., Taskin, E., Vaccari, F., Soldano, M., Piccinini, S., Frache, A., Remelli, S., Menta, C., Sandro Cocconcelli, P., Puglisi, E., 2022a. Anaerobic digestion and aerobic composting of rigid biopolymers in bio-waste treatment: fate and effects on the final compost. Bioresour. Technol. 351, 126934. https://doi.org/10.1016/j.biortech.2022.126934.
Bandini, F., Vaccari, F., Soldano, M., Piccinini, S., Misci, C., Bellotti, G., Taskin, E., Cocconcelli, P.S., Puglisi, E., 2022b. Rigid bioplastics shape the microbial communities involved in the treatment of the organic fraction of municipal solid waste. Front. Microbiol. 13, 1–18. https://doi.org/10.3389/fmicb.2022.1035561.
Barik, D., 2018. Energy extraction from toxic waste originating from food processing industries. Energy from Toxic Organic Waste for Heat and Power Generation. Elsevier Ltd. https://doi.org/10.1016/B978-0-08-102528-4.00003-1.
Bátori, V., Åkesson, D., Zamani, A., Taherzadeh, M.J., Sárvári Horváth, I., 2018. Anaerobic degradation of bioplastics: a review. Waste Manag. 80, 406–413. https://doi.org/10.1016/j.wasman.2018.09.040.
Battista, F., Frison, N., Bolzonella, D., 2021. Can bioplastics be treated in conventional anaerobic digesters for food waste treatment? Environ. Technol. Innov. 22, 101393. https://doi.org/10.1016/j.eti.2021.101393.
Benn, N., Zitomer, D., 2018. Pretreatment and anaerobic co-digestion of selected PHB and PLA bioplastics. Front. Environ. Sci. 5, 1–9. https://doi.org/10.3389/fenvs.2017.00093.
Bernal, M.P., Alburquerque, J.A., Moral, R., 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresour. Technol. 100, 5444–5453. https://doi.org/10.1016/j.biortech.2008.11.027.
Bourgogne, R.M. De, 2021. The Market for Biogas Plants in Japan and Opportunities for EU Companies, pp. 1–50.
CADER, 2020. Biometano como complemento del gas natural.
Cazaudehore, G., Guyoneaud, R., Evon, P., Martin-Closas, L., Pelacho, A.M., Raynaud, C., Monlau, F., 2022a. Can anaerobic digestion be a suitable end-of-life scenario for biodegradable plastics? A critical review of the current situation, hurdles, and challenges. Biotechnol. Adv. 56, 107916. https://doi.org/10.1016/j.biotechadv.2022.107916.
Cazaudehore, G., Guyoneaud, R., Lallement, A., Gassie, C., Monlau, F., 2022b. Biochemical methane potential and active microbial communities during anaerobic digestion of biodegradable plastics at different inoculum-substrate ratios. J Environ Manage 324, 116369. https://doi.org/10.1016/j.jenvman.2022.116369.
Cazaudehore, G., Monlau, F., Gassie, C., Lallement, A., Guyoneaud, R., 2023. Active microbial communities during biodegradation of biodegradable plastics by mesophilic and thermophilic anaerobic digestion. J. Hazard Mater. 443, 130208. https://doi.org/10.1016/j.jhazmat.2022.130208.
CCAC, 2021. Chile and Canada Partner to Reduce Emissions from the Waste Management Sector.
CIBiogas, 2023. Biogas map. https://app.powerbi.com/view?r=eyJrIjoiODc2NThhOGItOTc2Ny00ZDc1LWI5MTMtYjYwZTRlYjFiOWQ3IiwidCI6ImMzOTg3ZmI3LTQ5ODMtNDA2Ny1iMTQ2LTc3MGU5MWE4NGViNSJ9&pageName=ReportSection6ed365e9760a3c113b0d, 6.19.23.
Clagnan, E., Cucina, M., Vilas Sajgule, R., De Nisi, P., Adani, F., 2023. Microbial community acclimatization enhances bioplastics biodegradation and biogas production under thermophilic anaerobic digestion. Bioresour. Technol. 390, 129889. https://doi.org/10.1016/j.biortech.2023.129889.
Crema, A., Dinelli, E., Fabbri, E., Galletti, P., Greggio, N., Lastella, V., Parodi, A., Pasteris, A., Pedrizzi, M., Samorì, C., 2024. Additives in bioplastics: chemical characterization, migration in water and effects on photosynthetic organisms. Sci. Total Environ. 955, 177205. https://doi.org/10.1016/j.scitotenv.2024.177205.
Cucina, M., de Nisi, P., Tambone, F., Adani, F., 2021a. The role of waste management in reducing bioplastics' leakage into the environment: a review. Bioresour. Technol. 337, 125459. https://doi.org/10.1016/j.biortech.2021.125459.
Cucina, M., De Nisi, P., Trombino, L., Tambone, F., Adani, F., 2021b. Degradation of bioplastics in organic waste by mesophilic anaerobic digestion, composting and soil incubation. Waste Manag. 134, 67–77. https://doi.org/10.1016/j.wasman.2021.08.016.
Cucina, M., Soggia, G., De Nisi, P., Giordano, A., Adani, F., 2022. Assessing the anaerobic degradability and the potential recovery of biomethane from different biodegradable bioplastics in a full-scale approach. Bioresour. Technol. 354, 127224. https://doi.org/10.1016/j.biortech.2022.127224.
Danish Energy Agency, & Institute of Engineering UNAM, 2017. Database of potential substrates for biogas generation in Mexico, p. 116.
De Bernardi, A., Bandini, F., Marini, E., Tagliabue, F., Casucci, C., Brunetti, G., Vaccari, F., Bellotti, G., Tabaglio, V., Fiorini, A., Ilari, A., Gnoffo, C., Frache, A., Taskin, E., Rossa, U.B., Ricardo, E.S.L., Martins, A.O., Duca, D., Puglisi, E.,

Pedretti, E.F., Vischetti, C., 2024. Integrated assessment of the chemical, microbiological and ecotoxicological effects of a bio-packaging end-of-life in compost. Sci. Total Environ. 951, 175403. https://doi.org/10.1016/j.scitotenv.2024.175403.
De Schueler, A., Mahler, C.F., 2003. Composting in Brazil. Bioremediation and Leachate Treatment 60, 50.
Dilkes-Hoffman, L., Ashworth, P., Laycock, B., Pratt, S., Lant, P., 2019. Public attitudes towards bioplastics – knowledge, perception and end-of-life management. Resour. Conserv. Recycl. 151, 104479. https://doi.org/10.1016/j.resconrec.2019.104479.
Durkin, A., Taptygin, I., Kong, Q., Gunam Resul, M.F.M., Rehman, A., Fernández, A.M.L., Harvey, A.P., Shah, N., Guo, M., 2019. Scale-up and sustainability evaluation of biopolymer production from citrus waste offering carbon capture and utilisation pathway. ChemistryOpen 8, 668–688. https://doi.org/10.1002/open.201900015.
EBA, 2023. BIOMETHANE MAP INFRASTRUCTURE FOR BIOMETHANE PRODUCTION.
EBA, 2022. Statistical Report 2022 Tracking Biogas and Biomethane.
Ebenezer, A.V., Kumar, M.D., Kavitha, S., Uan, D.K., Banu, J.R., 2020. State of the art of food waste management in various countries. Food Waste to Valuable Resources: Applications and Management. INC. https://doi.org/10.1016/B978-0-12-818353-3.00014-6
ECN, 2023a. Country Report 2023 Germany.
ECN, 2023b. Country Report 2023 Italy.
ECN, 2023c. Country Report 2023 Belgium (Flanders).
ECN, 2023d. Country Report 2023 Austria.
ECN, 2022a. Position Paper on the Role of Organic Waste Derived Soil Improvers and Organic Fertilizers within Carbon Farming Initiative.
ECN, 2022b. Summary of ECN ` S 2022 Data Report : 20 Years of the European Compost Network and Biowaste Management in Europe ECN Summary Report.
ECN, 2020a. Country Report Denmark. Harmonisierung strafrechtlicher Sanktionen in der Europäischen Union 1001, 77–108. https://doi.org/10.5771/9783748902102-77.
ECN, 2020b. ECN Country Report 2019 Austria 2016, pp. 2016–2017.
ECN, 2019. ECN Country Report: Finland 4.
ECN, 2018. ECN Country Report 2018 Netherlands.
ECN, 2017a. ECN Country Report 2017 Norway.
ECN, 2017b. ECN Country Report 2015 Sweden 4.
Emadian, S.M., Onay, T.T., Demirel, B., 2017. Biodegradation of bioplastics in natural environments. Waste Manag. 59, 526–536. https://doi.org/10.1016/j.wasman.2016.10.006.
European Bioplastics, 2023. Bioplastics. https://doi.org/10.1021/cen-v050n023.p003.
European Commission, 2022. "EU policy framework on biobased, biodegradable and compostable plastics" COM (2022) 682.
European Compost Network, 2022. Compost and digestate for a circular bioeconomy: overview of bio-waste collection, treatment & markets across Europe. Ecn Data Report, 2022.
Fallis, A.G., 2013. Biomass to renewable energy processes. J. Chem. Inf. Model. https://doi.org/10.1017/CBO9781107415324.004.
Falzarano, M., Marìn, A., Cabedo, L., Polettini, A., Pomi, R., Rossi, A., Zonfa, T., 2024. Alternative end-of-life options for disposable bioplastic products: degradation and ecotoxicity assessment in compost and soil. Chemosphere 362, 142648. https://doi.org/10.1016/j.chemosphere.2024.142648.
FAO, 2011. Manual del Biogás 120, 120. http://www.fao.org/docrep/019/as400s/as400s.pdf.
Fredi, G., Dorigato, A., 2021. Recycling of bioplastic waste: a review. Advanced Industrial and Engineering Polymer Research 4, 159–177. https://doi.org/10.1016/j.aiepr.2021.06.006.
Gadaleta, G., De Gisi, S., Chong, Z.K., Heerenklage, J., Notarnicola, M., Kuchta, K., Cafiero, L., Oliviero, M., Sorrentino, A., Picuno, C., 2023. Degradation of thermoplastic cellulose acetate-based bioplastics by full-scale experimentation of industrial anaerobic digestion and composting. Chem. Eng. J. 462, 142301. https://doi.org/10.1016/j.cej.2023.142301.
Gadaleta, G., De Gisi, S., Picuno, C., Heerenklage, J., Cafiero, L., Oliviero, M., Notarnicola, M., Kuchta, K., Sorrentino, A., 2022. The influence of bio-plastics for food packaging on combined anaerobic digestion and composting treatment of organic municipal waste. Waste Manag. 144, 87–97. https://doi.org/10.1016/j.wasman.2022.03.014.
García-Depraect, O., Bordel, S., Lebrero, R., Santos-Beneit, F., Börner, R.A., Börner, T., Muñoz, R., 2021. Inspired by nature: microbial production, degradation and valorization of biodegradable bioplastics for life-cycle-engineered products. Biotechnol. Adv. 53. https://doi.org/10.1016/j.biotechadv.2021.107772.
Garcia-Depraect, O., Lebrero, R., Martínez-Mendoza, L.J., Rodriguez-Vega, S., Aragão Börner, R., Börner, T., Munoz, R., 2023. Enhancement of biogas production rate from bioplastics by alkaline pretreatment. Waste Manag. 164, 154–161. https://doi.org/10.2139/ssrn.4312721.
García-Depraect, O., Lebrero, R., Rodriguez-Vega, S., Bordel, S., Santos-Beneit, F., Martínez-Mendoza, L.J., Aragão Börner, R., Börner, T., Muñoz, R., 2022. Biodegradation of bioplastics under aerobic and anaerobic aqueous conditions: kinetics, carbon fate and particle size effect. Bioresour. Technol. 344. https://doi.org/10.1016/j.biortech.2021.126265.
García-Depraect, O., Martínez-Mendoza, L.J., Aragão Börner, R., Zimmer, J., Muñoz, R., 2024. Biomethanization of rigid packaging made entirely of poly(3-hydroxybutyrate-co-3-hydroxyhexanoate): mono- and co-digestion tests and microbial insights. Bioresour. Technol. 408, 131180. https://doi.org/10.1016/j.biortech.2024.131180.
Ghasemlou, M., Barrow, C.J., Adhikari, B., 2024. The future of bioplastics in food packaging: an industrial perspective. Food Packag. Shelf Life 43, 101279. https://doi.org/10.1016/j.fpsl.2024.101279.
Ghassan Alsultan, A., Asikin-Mijan, N., Kareem Obeas, L., Isalam, A., Mansir, N., Fawzi Nassar, M., Zulaika Razali, S., Yunus, R., Hin Taufiq-Yap, Y., 2023. Biofuel and biorefinery technologies, biochar - productive technologies, properties and applications. https://doi.org/10.5772/intechopen.104984.
Ghorpade, V.M., Gennadios, A., Hanna, M.A., 2001. Laboratory composting of extruded poly(lactic acid) sheets. Bioresour. Technol. 76, 57–61. https://doi.org/10.1016/S0960-8524(00)00077-8.
Gunes, B., Stokes, J., Davis, P., Connolly, C., Lawler, J., 2019. Pre-treatments to enhance biogas yield and quality from anaerobic digestion of whiskey distillery and brewery wastes: a review. Renew. Sustain. Energy Rev. 113, 109281. https://doi.org/10.1016/J.RSER.2019.109281.
Guo, J., Peng, Y., Ni, B.J., Han, X., Fan, L., Yuan, Z., 2015. Dissecting microbial community structure and methane-producing pathways of a full-scale anaerobic reactor digesting activated sludge from wastewater treatment by metagenomic sequencing. Microb Cell Fact 14, 1–11. https://doi.org/10.1186/s12934-015-0218-4.
Guo, X., Liu, H., Wu, S., 2019. Humic substances developed during organic waste composting: formation mechanisms, structural properties, and agronomic functions. Sci. Total Environ. 662, 501–510. https://doi.org/10.1016/j.scitotenv.2019.01.137.
Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E., Purnell, P., 2018. An overview of chemical additives present in plastics: migration, release, fate and environmental impact during their use, disposal and recycling. J. Hazard Mater. 344, 179–199. https://doi.org/10.1016/j.jhazmat.2017.10.014.
Hobbs, S.R., Harris, T.M., Barr, W.J., Landis, A.E., 2021. Life cycle assessment of bioplastics and food waste disposal methods. Sustainability 13. https://doi.org/10.3390/su13126894.
Holland Circular Hotspot, 2021. Waste Management Country Report: Argentina.
Holland circular hotspot HCH, 2021. Waste Management Country Report. México.
Hou, P., Xu, Y., Taiebat, M., Lastoskie, C., Miller, S.A., Xu, M., 2018. Life cycle assessment of end-of-life treatments for plastic film waste. J. Clean. Prod. 201, 1052–1060. https://doi.org/10.1016/j.jclepro.2018.07.278.
Huang, C., Wang, W., Sun, X., Shen, J., Wang, L., 2020. A novel acetogenic bacteria isolated from waste activated sludge and its potential application for enhancing anaerobic digestion performance. J Environ Manage 255, 109842. https://doi.org/10.1016/J.JENVMAN.2019.109842.
Idris, S.N., Amelia, T.S.M., Bhubalan, K., Lazim, A.M.M., Zakwan, N.A.M.A., Jamaluddin, M.I., Santhanam, R., Amirul, A.A.A., Vigneswari, S., Ramakrishna, S., 2023. The degradation of single-use plastics and commercially viable bioplastics in the environment: a review. Environ. Res. 231, 115988. https://doi.org/10.1016/j.envres.2023.115988.
IEA, 2022. Energy Policy Review Hungary.
Im, S., Hwang, I., Weonjae, K., Kim, D.-H., Kang, J.-H., Kang, S., 2024. Enhancing methane production potential of biodegradable plastics by hydrothermal pretreatment. Environ. Technol. Innov. 34, 103599. https://doi.org/10.1016/j.eti.2024.103599.
IRENA, 2021. Renewable Capacity Statistiques.
Islam, M., Xayachak, T., Haque, N., Lau, D., Bhuiyan, M., Pramanik, B.K., 2024. Impact of bioplastics on environment from its production to end-of-life. Process Saf. Environ. Prot. 188, 151–166. https://doi.org/10.1016/j.psep.2024.05.113.
Jung, H., Shin, G., Kwak, H., Hao, L.T., Jegal, J., Kim, H.J., Jeon, H., Park, J., Oh, D.X., 2023. Review of polymer technologies for improving the recycling and upcycling efficiency of plastic waste. Chemosphere 320, 138089. https://doi.org/10.1016/j.chemosphere.2023.138089.
Kalita, N.K., Damare, N.A., Hazarika, D., Bhagabati, P., Kalamdhad, A., Katiyar, V., 2021. Biodegradation and characterization study of compostable PLA bioplastic containing algae biomass as potential degradation accelerator. Environmental Challenges 3, 100067. https://doi.org/10.1016/j.envc.2021.100067.
Kang, J.H., Kang, S.W., Kim, W.J., Kim, D.H., Im, S.W., 2022. Anaerobic Co-digestion of bioplastics and food waste under mesophilic and thermophilic conditions: synergistic effect and biodegradation. Fermentation 8. https://doi.org/10.3390/fermentation8110638.
Kawai, F., 2010. Polylactic acid (PLA)-degrading microorganisms and PLA depolymerases. ACS (Am. Chem. Soc.) Symp. Ser. 1043, 405–414. https://doi.org/10.1021/bk-2010-1043.ch027.
Kawai, Kosuke, Liu, Chen, Premakumara Jagath, 2020. CCET guideline series on intermediate municipal solid waste treatment technologies. Composting 47.
Kirchmeyer, F., Stürmer, B., Hofmann, F., Sainz Arnau, A., 2020. Categorization of European Biogas Technologies 1–127.
Knoll, M., Hamm, T.M., Wagner, F., Martinez, V., Pleiss, J., 2009. The PHA Depolymerase Engineering Database: a systematic analysis tool for the diverse family of polyhydroxyalkanoate (PHA) depolymerases. BMC Bioinf. 10, 1–8. https://doi.org/10.1186/1471-2105-10-89.
Kosheleva, A., Gadaleta, G., De Gisi, S., Heerenklage, J., Picuno, C., Notarnicola, M., Kuchta, K., Sorrentino, A., 2023. Co-digestion of food waste and cellulose-based bioplastic: from batch to semi-continuous scale investigation. Waste Manag. 156, 272–281. https://doi.org/10.1016/j.wasman.2022.11.031.
Kriswantoro, J.A., Chu, C.-Y., Chang, T.-R., Pai, H.-J., Chang, C.-K., Chiu, Y.-P., 2023. Comparison of thermal alkaline pretreatment and zinc acetate-catalyzed methanolysis (MtOH-ZnOAc) for anaerobic digestion of bioplastic waste. Bioresour. Technol. 377, 128959. https://doi.org/10.1016/j.biortech.2023.128959.
Lavagnolo, M.C., Ruggero, F., Pivato, A., Boaretti, C., Chiumenti, A., 2020. Composting of starch-based bioplastic bags: small scale test of degradation and size reduction trend. Detritus 12, 57–65. https://doi.org/10.31025/2611-4135/2020.14008.
Li, Z., Lin, H., Ishii, N., Chen, G.-Q., Inoue, Y., 2007. Study of enzymatic degradation of microbial copolyesters consisting of 3-hydroxybutyrate and medium-chain-length 3-

hydroxyalkanoates. Polym Degrad Stab 92, 1708–1714. https://doi.org/10.1016/j.polymdegradstab.2007.06.001.
Lin, L., Xu, F., Ge, X., Li, Y., 2019. Biological treatment of organic materials for energy and nutrients production—anaerobic digestion and composting. Advances in Bioenergy, first ed. Elsevier Inc. https://doi.org/10.1016/bs.aibe.2019.04.002.
Liu, Q., He, X., Wang, K., Li, D., 2023. Biochar drives humus formation during composting by regulating the specialized metabolic features of microbiome. Chem. Eng. J. 458, 141380. https://doi.org/10.1016/j.cej.2023.141380.
Liu, T., Chen, H., Zhou, Y., Awasthi, S.K., Qin, S., Liu, H., Zhang, Z., Pandey, A., Varjani, S., Awasthi, M.K., 2022. Composting as a sustainable technology for integrated municipal solid waste management. Biomass, Biofuels. Elsevier Inc, Biochemicals. https://doi.org/10.1016/b978-0-323-88511-9.00002-1.
Liu, X., Yan, Z., Yue, Z.B., 2011. Biogas. Comprehensive Biotechnology, second ed. 3, pp. 99–114. https://doi.org/10.1016/B978-0-08-088504-9.00165-3.
Liu, Z., Wang, X., Wang, F., Bai, Z., Chadwick, D., Misselbrook, T., Ma, L., 2020. The progress of composting technologies from static heap to intelligent reactor: benefits and limitations. J. Clean. Prod. 270, 122328. https://doi.org/10.1016/j.jclepro.2020.122328.
Lu, J., Qiu, Y., Muhmood, A., Zhang, L., Wang, P., Ren, L., 2023. Appraising co-composting efficiency of biodegradable plastic bags and food wastes: assessment microplastics morphology, greenhouse gas emissions, and changes in microbial community. Sci. Total Environ. 875, 162356. https://doi.org/10.1016/j.scitotenv.2023.162356.
Martín-Lara, M.A., Moreno, J.A., Garcia-Garcia, G., Arjandas, S., Calero, M., 2022. Life cycle assessment of mechanical recycling of post-consumer polyethylene flexible films based on a real case in Spain. J. Clean. Prod. 365, 132625. https://doi.org/10.1016/j.jclepro.2022.132625.
Mat Yasin, N., Akkermans, S., Van Impe, J.F.M., 2022. Enhancing the biodegradation of (bio)plastic through pretreatments: a critical review. Waste Manag. 150, 1–12. https://doi.org/10.1016/j.wasman.2022.06.004.
Nayanathara Thathsarani Pilapitiya, P.G.C., Ratnayake, A.S., 2024. The world of plastic waste: a review. Cleaner Materials 11, 100220. https://doi.org/10.1016/j.clema.2024.100220.
OECD, 2020. Municipal Waste by Treatment Operation, Percentage of Total Treatment in 2021.
Peng, W., Nie, R., Lü, F., Zhang, H., He, P., 2024. Biodegradability of PBAT/PLA coated paper and bioplastic bags under anaerobic digestion. Waste Manag. 174, 218–228. https://doi.org/10.1016/j.wasman.2023.11.037.
Peng, W., Wang, Z., Shu, Y., Lü, F., Zhang, H., Shao, L., He, P., 2022. Fate of a biobased polymer via high-solid anaerobic co-digestion with food waste and following aerobic treatment: insights on changes of polymer physicochemical properties and the role of microbial and fungal communities. Bioresour. Technol. 343, 126079. https://doi.org/10.1016/j.biortech.2021.126079.
Peng, X., Zhang, S., Li, L., Zhao, X., Ma, Y., Shi, D., 2018. Long-term high-solids anaerobic digestion of food waste: effects of ammonia on process performance and microbial community. Bioresour. Technol. 262, 148–158. https://doi.org/10.1016/J.BIORTECH.2018.04.076.
Plastics Europe, 2024. The Circular Economy for Plastics A European Analysis.
Pooja, N., Chakraborty, I., Rahman, M.H., Mazumder, N., 2023. An insight on sources and biodegradation of bioplastics: a review. 3 Biotech. https://doi.org/10.1007/s13205-023-03638-4.
Rajendran, K., Aslanzadeh, S., Taherzadeh, M.J., 2012. Household biogas digesters-A review, Energies. https://doi.org/10.3390/en5082911.
Ramírez-Higareda, B.L., Leyva-Huitrón, R., Chimil-Molina, R., López-Hernández, J.E., Harder, B., 2019. Biogas Plants in Denmark and Mexico, vol. 39.
Rasheed, T., Vattathurvalappil, S.H., Shaukat, M.M., Theravalappil, R., Ali, U., Ummer, A.C., Saleem, M.T. Bin, Jaseer, E.A., Imran, M., 2024. Recent updates on biodegradability and recyclability of bioplastics - towards a new era in sustainability. Sustainable Materials and Technologies 41, e01051. https://doi.org/10.1016/j.susmat.2024.e01051.
Regulation (EU) 2019/1009, 2019.
Rodero, M.R., Ángeles, R., Marín, D., Díaz, I., Colzi, A., Posadas, E., Lebrero, R., Muñoz, R., 2018. In: Tabatabaei, M., Ghanavati, H. (Eds.), Biogas Purification and Upgrading Technologies BT - Biogas: Fundamentals, Process, and Operation. Springer International Publishing, Cham, pp. 239–276. https://doi.org/10.1007/978-3-319-77335-3_10.
Rostkowski, K.H., Criddle, C.S., Lepech, M.D., 2012. Cradle-to-Gate life cycle assessment for a cradle-to-cradle cycle: biogas-to-bioplastic (and back). Environ. Sci. Technol. 46, 9822–9829. https://doi.org/10.1021/es204541w.
Ruggero, F., Belardi, S., Carretti, E., Lotti, T., Lubello, C., Gori, R., 2022. Rigid and film bioplastics degradation under suboptimal composting conditions: a kinetic study. Waste Manag. Res. 40, 1311–1321. https://doi.org/10.1177/0734242X211063731.
Ruggero, F., Carretti, E., Gori, R., Lotti, T., Lubello, C., 2020. Monitoring of degradation of starch-based biopolymer film under different composting conditions, using TGA, FTIR and SEM analysis. Chemosphere 246, 125770. https://doi.org/10.1016/j.chemosphere.2019.125770.
Scarlat, N., Dallemand, J.F., Fahl, F., 2018. Biogas: developments and perspectives in Europe. Renew. Energy 129, 457–472. https://doi.org/10.1016/j.renene.2018.03.006.
Schmidt, J.B.J.E., Processes, I.S., Conversion, B., 2019. Biorefinery Book.
Soni, R., Devi, S., 2022. Composting process: fundamental and molecular aspects. Relationship between Microbes and the Environment for Sustainable Ecosystem Services, Volume 1: Microbial Products for Sustainable Ecosystem Services. Elsevier Inc. https://doi.org/10.1016/B978-0-323-89938-3.00012-8.
Srisowmeya, G., Chakravarthy, M., Nandhini Devi, G., 2020. Critical considerations in two-stage anaerobic digestion of food waste – a review. Renew. Sustain. Energy Rev. 119, 109587. https://doi.org/10.1016/J.RSER.2019.109587.
Sun, Y., Ren, X., Rene, E.R., Wang, Z., Zhou, L., Zhang, Z., Wang, Q., 2021. The degradation performance of different microplastics and their effect on microbial community during composting process. Bioresour. Technol. 332, 125133. https://doi.org/10.1016/j.biortech.2021.125133.
Tayibi, S., Monlau, F., Bargaz, A., Jimenez, R., Barakat, A., 2021. Synergy of anaerobic digestion and pyrolysis processes for sustainable waste management: a critical review and future perspectives. Renew. Sustain. Energy Rev. 152, 111603. https://doi.org/10.1016/j.rser.2021.111603.
Tokiwa, Y., Calabia, B.P., Ugwu, C.U., Aiba, S., 2009. Biodegradability of plastics. Int. J. Mol. Sci. 10, 3722–3742. https://doi.org/10.3390/ijms10093722.
Tseng, H.C., Fujimoto, N., Ohnishi, A., 2019. Biodegradability and methane fermentability of polylactic acid by thermophilic methane fermentation. Bioresour. Technol. Rep. 8, 100327. https://doi.org/10.1016/j.biteb.2019.100327.
van der Linden, A., Reichel, A., 2020. Bio-waste in Europe: Turning Challenges into Opportunities, EEA Report No 04/2020.
Vargas-Estrada, L., Longoria, A., Arenas, E., Moreira, J., Okoye, P.U., Bustos-Terrones, Y., Sebastian, P.J., 2021. A review on current trends in biogas production from miroalgae biomass and microalgae waste by anaerobic digestion and Co-digestion. Bioenergy Res. https://doi.org/10.1007/s12155-021-10276-2.
Vaverková, M.D., Adamcová, D., 2015. Biodegrability of bioplastic materials in a controlled composting environment. Journal of Ecological Engineering 16, 155–160. https://doi.org/10.12911/22998993/2949.
Venkiteshwaran, K., Benn, N., Seyedi, S., Zitomer, D., 2019. Methane yield and lag correlate with bacterial community shift following bioplastic anaerobic co-digestion. Bioresour. Technol. Rep. 7, 100198. https://doi.org/10.1016/j.biteb.2019.100198.
Wang, L., Chang, R., Ren, Z., Meng, X., Li, Y., Gao, M., 2024. Mature compost promotes biodegradable plastic degradation and reduces greenhouse gas emission during food waste composting. Sci. Total Environ. 926, 172081. https://doi.org/10.1016/j.scitotenv.2024.172081.
Wang, M., Li, Y., Peng, H., Wang, J., Li, Q., Li, P., Fan, J., Liu, S., Zheng, G., 2023. Review: biotic and abiotic approaches to artificial humic acids production. Renew. Sustain. Energy Rev. 187, 113771. https://doi.org/10.1016/j.rser.2023.113771.
Xing, R., Sun, H., Du, X., Lin, H., Qin, S., Chen, Z., Zhou, S., 2023. Enhanced degradation of microplastics during sludge composting via microbially-driven Fenton reaction. J. Hazard Mater. 449, 131031. https://doi.org/10.1016/j.jhazmat.2023.131031.
Xu, M., Sun, H., Chen, E., Yang, M., Wu, C., Sun, X., Wang, Q., 2023. From waste to wealth: innovations in organic solid waste composting. Environ. Res. 229, 115977. https://doi.org/10.1016/j.envres.2023.115977.
Yagi, H., Ninomiya, F., Funabashi, M., Kunioka, M., 2014. Mesophilic anaerobic biodegradation test and analysis of eubacteria and archaea involved in anaerobic biodegradation of four specified biodegradable polyesters. Polym Degrad Stab 110, 278–283. https://doi.org/10.1016/j.polymdegradstab.2014.08.031.
Yagi, H., Ninomiya, F., Funabashi, M., Kunioka, M., 2013. Thermophilic anaerobic biodegradation test and analysis of eubacteria involved in anaerobic biodegradation of four specified biodegradable polyesters. Polym Degrad Stab 98, 1182–1187. https://doi.org/10.1016/j.polymdegradstab.2013.03.010.
Yang, X., Nie, J., Wang, D., Zhao, Z., Kobayashi, M., Adachi, Y., Shimizu, K., Lei, Z., Zhang, Z., 2019. Enhanced hydrolysis of waste activated sludge for methane production via anaerobic digestion under N2-nanobubble water addition. Science of The Total Environment 693, 133524. https://doi.org/10.1016/J.SCITOTENV.2019.07.330.
Yu, C., Dongsu, B., Tao, Z., Xintong, J., Ming, C., Siqi, W., Zheng, S., Yalei, Z., 2023. Anaerobic co-digestion of three commercial bio-plastic bags with food waste: effects on methane production and microbial community structure. Sci. Total Environ. 859, 159967. https://doi.org/10.1016/j.scitotenv.2022.159967.
Yu, J., Plackett, D., Chen, L.X.L., 2005. Kinetics and mechanism of the monomeric products from abiotic hydrolysis of poly[(R)-3-hydroxybutyrate] under acidic and alkaline conditions. Polym Degrad Stab 89, 289–299. https://doi.org/10.1016/j.polymdegradstab.2004.12.026.
Zaborowska, M., Bernat, K., Pszczółkowski, B., Cydzik-Kwiatkowska, A., Kulikowska, D., Wojnowska-Baryła, I., 2023. Multi-faceted analysis of thermophilic anaerobic biodegradation of poly(lactic acid)-based material. Waste Manag. 155, 40–52. https://doi.org/10.1016/j.wasman.2022.10.031.