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Review

Municipal Solid Waste as a Renewable Energy Source: Advances in Thermochemical Conversion Technologies and Environmental Impacts

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Abstract: This review examines the potential of municipal solid waste (MSW) as a renewable energy source, focusing on recent advances in thermochemical conversion technologies and their environmental impacts. The exponential growth of urban populations has led to a surge in MSW, necessitating sustainable waste management solutions. Traditional disposal methods, such as landfilling and incineration, have significant environmental drawbacks. However, advancements in waste-to-energy (WtE) technologies, including incineration, pyrolysis, and gasification, offer promising alternatives for energy recovery and resource utilization. This review explores the composition of MSW, its classification as a renewable resource, and the thermochemical conversion technologies that transform waste into energy. The environmental impacts of these technologies, particularly emissions and air quality concerns, are critically analyzed. The review highlights the evolving regulatory landscape and the implementation of advanced emission reduction systems. The findings underscore the importance of integrating innovative waste management strategies to promote a circular economy and achieve sustainable development goals.

Keywords: municipal solid waste (MSW); renewable energy; waste-to-energy; incineration; pyrolysis; gasification

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1. Introduction

The 21st century is marked by unprecedented growth in the global population, leading to a surge in urbanization and changing consumption patterns. As cities expand and consumer habits intensify, the generation of municipal solid waste (MSW) increases proportionally. According to the United Nations Department of Economic and Social Affairs, the world's population, currently surpassing 7.8 billion, is projected to reach nearly 9.7 billion by 2050 [1]. This raises the question about the limits of human population growth. Resources such as water, arable land, and fossil fuels are not unlimited. Moreover, the growth of the human population is associated with increased environmental pollution. According to World Bank reports, global waste generation is expected to increase from 2.01 billion tonnes per year in 2020 to 3.40 billion tonnes by 2050. This represents a 70% increase, which is more than twice the anticipated rate of population growth over the same period [2,3]. The escalating volume of waste not only strains existing disposal and treatment facilities but also exacerbates the environmental, economic, and social challenges associated with waste management [4]. Moreover, conventional methods of waste disposal, such as landfilling and basic incineration, involve the wastage of non-renewable resources, such as in the form of plastic packaging. This situation necessitates a shift towards more sustainable and innovative waste management strategies—towards the utilization of municipal waste as a resource. From an environmental protection standpoint, this approach embodies the essence of sustainable development.

In this review, we explore how municipal solid waste, traditionally viewed as a burdensome byproduct of urban life, is increasingly recognized as a resource. This paradigm shift is supported by advancements in waste conversion technologies that enable the sustainable extraction of energy and materials from waste, thereby contributing to a circular economy. The following sections will delve into the composition of MSW, its potential as a renewable energy source, and the technologies transforming waste management practices.

2. Fundamentals of Municipal Solid Waste for Energy Generation

2.1. MSW as a Renewable Energy Source

In 1972, Donella H. Meadows et al., in their influential work "*The Limits to Growth*" [5], pointed out the consequences of long-term population growth and industrialization, including the exponential depletion of non-renewable natural resources. This was not, of course, the first work of this type—as early as 1798, Thomas Malthus had drawn attention to the problem of geometric population growth versus the linear growth of available food resources [6]. However, the work of [5] led to a worldwide debate on the limits of human population growth and the potential economic problems resulting from the depletion of available fossil resources. A confirmation of these observations was the oil crisis of 1973, which led to a global recession and increased unemployment. In countries dependent on oil imports, energy consumption restrictions were introduced, leading to increased living costs. However, the oil crisis became a catalyst for intensified research on the development of renewable energy sources (RESs), particularly in the crisis-hit United States and Western European countries. This research focused primarily on the technological development of wind and water turbines. In 1980, the first commercial wind farm, consisting of 30 wind turbines with a capacity of 30 kW each, was launched in Crotched Mountain, New Hampshire [7].

One of the key moments in the development of RESs in Europe was the publication of the "White Paper on Renewable Energy" entitled "Energy for the Future: Renewable Energy Sources" by the European Union in November 1997 [8]. This document aimed to outline policies and strategies for promoting renewable energy sources in Europe, which led to a broad understanding and acceptance of the term. The share of RESs in Europe in 1997 was about 6% [9], while the strategic goal was to achieve 12% by the end of 2010. According to data from the European Environment Agency, the strategic goals were achieved at a level of 12.5% [10]. The "White Paper on Renewable Energy" emphasized the importance of using municipal waste as an energy source and the need for a regulatory and planning approach by the EU to waste management in the context of energy production. Although the possibility of using municipal waste for energy production (WtE) was known much earlier [11], the publication [8] defined biomass from municipal waste as a renewable energy source. This issue will be significant in the future when trying to classify municipal waste as renewable resources.

As noted by Rybár et al. [12]—the basis for dividing energy sources into renewable and non-renewable is the criterion of the "renewability" or "exhaustibility" of the energy source. A key feature of renewable resources is their ability to regenerate through natural processes in a relatively short period compared to non-renewable resources, such as fossil fuels. Although the definition of renewable energy sources has appeared over the years in various scientific and political contexts (e.g., [13–22]), serious discussions on classifying MSW as renewable energy began only in the early 21st century, among major energy and environmental agencies. Interestingly, these classifications have often been contradictory, even within the same publication. For example, the definition of renewable energy in Section 203 of the 2005 Energy Policy Act [23] explicitly included electricity derived from MSW as a "renewable energy" resource qualifying for meeting the federal renewable energy purchase requirement specified in this section. However, many other sections of the same act did not include MSW as a renewable energy source qualifying for the purposes of programs aimed at developing, evaluating, or supporting renewable energy. In response to these inconsistencies, the U.S. Energy Information Administration (EIA) published a report in 2007 thoroughly analyzing this issue [24]. According to this report, MSW can be considered a renewable energy source only to the extent that the energy content of the MSW stream comes from biogenic components. The non-biogenic part of MSW is reported as "Other non-renewable waste". Ultimately, it can be concluded that the classification of MSW as renewable resources in the United States and Europe has a similar character and refers to the biomass–organic fraction–biogenic part of MSW. It should be emphasized that according to the EIA [24], the non-biogenic part of municipal waste includes morphological components such as plastics and rubber, which themselves are an effective energy source, and thus the energy derived from them will be treated as non-renewable.

Currently, the classification of municipal waste as renewable resources is not a closed issue. The statutory definition of renewable resources, ratified by 108 members of the International Renewable Energy Agency (IRENA) [21], states that "renewable energy includes all forms of energy produced from renewable sources in a sustainable manner, INCLUDING bioenergy, geothermal energy, hydropower, ocean energy, solar energy, and wind energy". As of the writing of this work, in no adopted definition of renewable resources has the list of these resources been definitively disclosed.

2.2. Composition and Properties of MSW

The global morphological composition of municipal solid waste varies depending on income levels and regions, reflecting differences in consumption, processing technologies, and waste management systems. World Bank studies, conducted for 217 countries and economies [2], show that high-income countries (with a GNI per capita of USD 13,846 or more in the fiscal year 2024 [25]) generate municipal waste with the smallest share of organic waste (32%). In other countries, the share of organic waste is above 50%. These results are fully consistent with the literature reports, e.g., [26–30]. The data are presented in Table 1.

Table 1. Global changes in morphological composition of municipal solid waste by income level [2].

Category	**Low-Income Countries**	**Lower-Middle-Income Countries**	**Upper-Middle-Income Countries**	**High-Income Countries**
Organic Waste	56%	54%	53%	32%
Plastic	6.4%	12%	12.5%	12%
Paper and Cardboard	27%	15%	17%	25%
Metal	1%	2%	3%	5%
Glass	2%	4%	2%	4%
Others	7%	13%	12.5%	22%

The diversity in waste composition across income levels underscores the importance of customized approaches to waste management that can effectively address the unique challenges and opportunities presented by varying waste streams. In low-income countries, where organic waste predominates, strategies focusing on composting and biogas production could be particularly beneficial. A comprehensive study of [31] shows that developed countries focus on reducing, reusing, and recycling MSW, whereas low-income countries struggle with inadequate waste infrastructure and informal waste sectors. Moreover, the effectiveness of waste management practices is heavily influenced by the legal and regulatory frameworks in place within different regions. In the European Union, the Waste Framework Directive (2008/98/EC) mandates that member states implement policies to prioritize waste prevention, recycling, and recovery. This directive also sets specific targets for recycling and the diversion of waste from landfills, fostering a more structured approach to waste management. In contrast, in many low-income countries, regulatory frameworks are either underdeveloped or poorly enforced, leading to significant challenges in waste collection and segregation. These countries often lack the necessary infrastructure, funding, and technical expertise to implement effective waste management policies, resulting in reliance on informal waste sectors and widespread environmental and public health issues [31,32].

In countries like Japan, strict regulations on waste segregation and recycling, such as the Containers and Packaging Recycling Law, have led to high recycling rates and a reduction in landfill use. However, despite the overall effectiveness of these regulations, there are some unresolved challenges related to the recycling of certain types of plastic packaging, which affect the efficiency of the entire system [33]. The United States, on the other hand, exhibits a more decentralized approach to waste management, with regulations varying significantly by state and municipality. While some states have robust recycling programs and stringent landfill diversion goals, others lag in implementing comprehensive waste management strategies. These regulatory differences contribute to the varying levels of success in waste management across regions, with high-income countries generally having more advanced systems in place. Effective regulation not only facilitates better waste segregation and collection but also encourages the development of infrastructure necessary for advanced waste-processing technologies, such as incineration and gasification. The data summary of [2] indicates that the areas with more advanced waste management systems are North America, Europe, and Central Asia (Table 2).

Table 2. Global changes in morphological composition of municipal solid waste by region [2].

Category	North America	Europe and Central Asia	Middle East and North Africa	South Asia	Sub-Saharan Africa
Organic Waste	<30%	36%	58%	64%	57%
Plastic	12%	18.6%	13%	9%	10%
Paper and Cardboard	28%	21%	8%	8%	9%
Metal	-	3%	3%	2%	4%
Glass	-	8%	3%	2%	3%
Others	55%	11.5%	15%	15%	17%

In the analysis of municipal solid waste (MSW) a range of advanced research techniques are used in order to determine the physical and chemical properties of municipal waste [34]. An understanding of these properties directly affects waste processing and logistics costs. The most important physical properties of waste are moisture (i.e., the content of non-chemically bound water), bulk density, granulometry, and magnetism. From the perspective of using waste as an energy source, chemical properties also matter, such as calorific value, elemental composition, overall chemical reactivity, radioactivity, and the content of heavy metals and ash.

From a technological point of view, it is assumed that biomass moisture above 50–65% can reduce the stability of the combustion process, increase emissions of CO and volatile organic compounds, and during gasification, lead to lower heating values and higher tar concentrations [35–39]. The moisture content of waste is influenced by several factors. Generally, due to the high water content in fruits and vegetables, selectively collected kitchen waste has the highest moisture content, reaching up to 75% wt. [40]. According to the findings of [41], relatively high moisture content is also characteristic of wood waste (>40% wt), paper, and textiles (<20% wt). Such materials as PE, PP, PS, and rubber have minimal capacity to absorb moisture. Research of [42] indicates a relationship between waste moisture content and particle size. The authors found that smaller fractions of MSW generally have higher moisture content, which affects the thermal properties of the waste. One of the oldest and still used charts for assessing the combustible properties of waste in this context is the Tanner diagram from 1965. This diagram shows the influence of three variables on the ability of waste to sustain combustion: moisture content, organic matter content, ash content. All these variables are expressed on a wet basis (wb), allowing for an accurate representation of the actual conditions in which the waste is burned. The diagram was constructed using field data from actual MSW incinerators from various parts of the world [43]. The threshold values in this chart suggest that waste mixtures with an ash content of up to 60% (wb), moisture content of up to 50% (wb), and combustible material content of at least 25% can sustain self-combustion. The Tanner diagram was updated in a publication by Komilis et al. [44]. The authors undertook a verification of this classic chart by conducting a series of laboratory experiments using various organic substrates. The updated charts indicate that waste can sustain self-combustion with an organic matter content of 15% dry weight, provided the moisture content does not exceed 20% wet weight. Additionally, combustion is possible with a moisture content of up to 60% wet weight, if the organic matter content is at least 40% wet weight. The results of both works [43,44] are detailed in Figure 1.

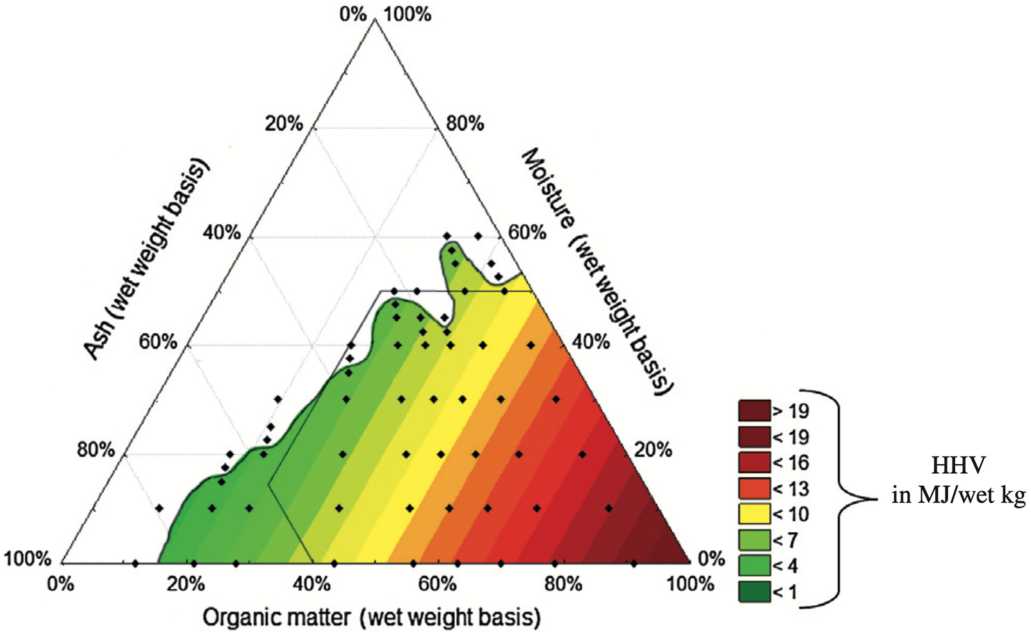

Figure 1. A triangular diagram showing the higher heating value as a function of the content of organic matter, moisture, and ash. The colored area indicates the region of self-sustained combustion. The polygon within the triangle represents the area of self-sustained combustion according to the Tanner diagram [43]. (Source: [44] with permission from Elsevier).

To assess the calorific value of waste, two values are customarily used: the higher heating value (HHV) and the lower heating value (LHV). The formal and precise definitions of HHV and LHV have evolved with advances in thermodynamics and chemical engineering. In a scientific and technical context, this terminology has been used since the 19th century, when thermodynamic theories were being developed and studies on fuel properties were being conducted. It is difficult to attribute these definitions to specific authors or dates, as they were the result of collective progress in the field of science and technology. Generally, the HHV is defined as the quantity of heat generated by complete combustion of a mass unit of sample at constant volume in an oxygen atmosphere, if both the water contained in the sample and that generated from the combined hydrogen remain in liquid form. If it is assumed that the water in the products remains in the form of steam, LHV can be calculated as the heat released by the combustion. One of the earlier works describing this relationship for municipal waste was the work of Franjo et al. [45], which can now be presented as:

$$\mathrm{LHV} = \mathrm{HHV}\ (1 - \mathrm{M}) - 2442\ (\mathrm{M} + 8.94\ \mathrm{H})\ [\mathrm{J/g}], \tag{1}$$

where 2442 is the heat of vaporization of water at 25 °C [J/g]; M and H are the corresponding values of moisture and hydrogen content in the sample of waste. It can therefore be assumed that an increase in moisture by 2% reduces the LHV by about 500 J/g. Hence, the application of biodrying waste before directing it to waste incineration increases its LHV.

The knowledge of the HHV of individual categories (morphological components) of municipal waste is crucial for the economic and technical analysis of energy recovery systems and the optimization of thermal processes such as combustion or pyrolysis. In a study [46], 252 experimental data points on the HHV of various waste categories were collected, providing a rich dataset for predictive analysis. According to the authors' analysis, plastic waste is characterized by one of the highest HHV values. This value can reach up to 46.6 MJ/kg, making plastic a valuable source of thermochemical energy. However, the high chlorine content in certain plastics, such as polyvinyl chloride (PVC), can reduce their energy value [41]. Leather and rubber waste can also be significant sources of energy in combustion processes and other energy recovery technologies. The HHV values of leather and rubber are among the highest in municipal waste, ranging from 19.67 to 42.14 MJ/kg. Paper and textiles, which are often components of municipal waste, have variable HHV values. The average HHV for paper is about 14.85 MJ/kg, while for textiles, it is 24.93 MJ/kg. These fractions have moderate energy values, making them effectively usable in energy recovery combustion processes. Organic waste, which has the lowest HHV and the highest proportion, significantly influences the overall HHV of mixed municipal waste, which averages 19.81 MJ/kg. The above experimental data collected by [46] are summarized in Table 3.

Table 3. Experimental data on chemical and calorific properties of various waste categories (based on [46]).

Category	Elemental Composition Range [% wt]					HHV [MJ/kg]	
	C	**H**	**N**	**S**	**O**	**HHV Range**	**Mean HHV**
Paper waste	30.50–59.18	3.46–9.25	0.05–2.9	0.0–1.50	27.5–46.9	10.40–19.30	14.85
Textile waste	46.19–66.85	5.34–9.72	0.1–4.62	0.0–0.40	18.7–43.6	18.69–31.16	24.93
Leather and rubber waste	42.01–84.22	5.18–9.44	0.0–10.0	0.0–2.46	2.67–23.3	19.67–42.14	30.91
MSW mixture	31.20–59.20	3.80–9.07	0.22–1.4	0.0–0.89	21.8–46.32	11.99–27.63	19.81
Plastic waste	38.00–92.00	4.90–14.2	0.0–6.26	0.0–0.86	0.05–48.62	15.82–46.60	31.21
Wood waste	36.20–53.30	4.75–6.66	0.05–6.99	0.02–1.2	30.34–47.7	14.61–23.26	18.94
Organic and other waste	20.62–50.94	2.87–7.60	0.5–6.26	0.0–0.81	4.0–48.62	9.10–22.61	15.86

3. Conversion Technologies

3.1. Thermochemical Conversion

Thermochemical conversion technologies offer a promising approach for transforming municipal solid waste (MSW) into renewable energy sources. These technologies primarily include incineration, pyrolysis, and gasification. Each of these processes involves high-temperature treatment of waste to produce energy-rich gases, oils, and solids, which can then be utilized for power generation and other applications.

3.1.1. Incineration

Currently, there are over 2700 waste-to-energy (WtE) incineration plants operating worldwide, with a total capacity to process approximately 530 million tons of waste annually. It is projected that, by 2032, the number of these facilities will exceed 3000, and their processing capacity will increase to over 700 million tons per year [47]. In Europe, there are around 500 WtE incineration plants, which process about 100 million tons of municipal, commercial, and industrial waste annually [48]. Incineration is one of the oldest and most established methods for MSW disposal and energy recovery [49]. The development of incineration plants began in the second half of the 19th century when cities like Manchester, Birmingham, and London started experimenting with various types of waste-burning furnaces. Initially, simple furnaces were used, which required the addition of coal to effectively burn the waste. In 1876, Alfred Fryer developed the first specialized furnace, known as the destructor, which was more efficient and could burn larger amounts of waste (Patent no. 3125) [50]. Over the following decades, various types of destructors were developed, becoming increasingly technologically advanced. The introduction of high-temperature furnaces, such as Horsfall's furnace, allowed for more efficient burning and better utilization of thermal energy for steam production. Horsfall's modifications included directing the flue gas channels through a red-hot reverberatory arch, which increased the efficiency of flue gas utilization. Due to its relatively high processing capacity (this furnace could burn from 8 to 10 tons of waste per day), Horsfall's design significantly improved waste management efficiency in many European cities [50]. At the beginning of the 20th century, incineration plants began to be combined with power plants, which allowed the use of heat generated in the waste incineration process for electricity production. In cities like London and Bradford, waste incineration plants were an integral part of the waste management and energy supply system.

As environmental awareness in society increased, waste incineration technology also evolved through the introduction of emission control systems. Over time, innovations such as filter bags, scrubbers, and electrostatic precipitators were implemented. Current technological challenges primarily involve reducing the emission of dioxins produced during waste incineration. Potential dioxin emissions from incineration plants are a common argument used during social protests. The term "dioxins" typically encompasses three groups of compounds: polychlorinated dibenzo-p-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), and polychlorinated biphenyls (PCBs) [51]. Since each benzene ring can have 0–4 hydrogen atoms replaced by chlorine atoms, the group of dioxins and furans includes 75 PCDD isomers and 135 PCDF isomers. One of the most toxic dioxins is 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). The lethal dose (LD50) for rats ranges from 10 µg/kg to 7.2 mg/kg body weight, depending on the strain [52]. There are no precise data on the lethal dose of TCDD in humans; however, studies conducted by Steenland et al. [53] showed that with a daily intake of TCDD at the level of 1.0 pg/kg body weight (twice the average background intake), the risk of cancer death increases by 0.05–0.9% above the background level of 12.4%. Toxicokinetic analysis based on TCDD concentrations in blood serum indicated that the half-life of TCDD in humans ranges from about 3 years to over 10 years, depending on the TCDD concentration in the serum [54].

Zhao et al. [51] describe three basic mechanisms of dioxin formation: high-temperature gas-phase reaction, "de novo" synthesis (first described de facto by Stieglitz and Vogg [55]), and precursor synthesis. High-temperature gas-phase reaction mainly occurs in the heat exchange zone of the combustion chamber at temperatures from 500 °C to 850 °C. Chlorine contained in the waste forms HCl and Cl_2, which then react to form dioxin precursors such as chlorobenzene and chlorophenol. "De novo" synthesis occurs at lower temperatures (around 300 °C). It involves the reaction of carbon, oxygen, chlorine, and catalysts such as Cu^{2+} on the surface of fly ash, leading to the formation of dioxins. Dioxins can also form from chlorinated organic compounds (e.g., chlorophenol, chlorobenzene) through homogeneous and heterogeneous catalytic reactions. Regardless of the dioxin formation mechanism, chlorine plays a key role, hence the need to limit its content in waste. According to Zheng and Qi [56], the chlorine content in municipal waste directed to incineration plants should not exceed 1.1%. One method of reducing chlorine content in waste is the optical-ballistic separation of PVC during the SRF fuel valorization stage [57].

The primary method for reducing dioxin emissions involves establishing appropriate combustion conditions, which include maintaining combustion temperatures above 1000 °C, ensuring a gas residence time in the combustion chamber longer than 1 s, and achieving a high degree of gas turbulence ($Re > 50{,}000$) [58]. It is also important to rapidly cool the flue gases from 400 to 250 °C. It has been demonstrated that, at high temperatures (>1600 °C), all organic compounds, including dioxins, are destroyed; however, during slow cooling of the gases below 500 °C, dioxin reformation occurs [59]. Additionally, to reduce dioxin emissions, inhibitors are added to the combustion process. The most commonly used dioxin formation inhibitors include sulfur compounds (such as ammonium sulfate, pyrite, and sodium thiosulfate), nitrogen compounds (such as ethanolamine, urea, and ammonia), and alkaline dechlorinators (such as quicklime and sodium hydroxide). These compounds work by transforming copper catalysts into less active forms or by converting Cl_2 to HCl, which significantly reduces dioxin formation. In recent years, modern flue-gas-cleaning technologies have also emerged. One of the most effective methods is the combination of activated carbon injection and bag filters (ACI + BF), which effectively removes dioxins from flue gases through adsorption on activated carbon. Another innovative method is the use of electrostatic precipitators with cooled electrodes (CE-ESP), which utilize the coupling of electrostatic forces, diffusio-phoresis, and thermophoresis to efficiently remove dioxins. The selective catalytic reduction (SCR) technology, typically associated with NOx removal, is also employed, where the V_2O_5-WO_3/TiO_2 catalyst effectively decomposes dioxins in the presence of ammonia [51]. In studies conducted at municipal waste incineration plants, it has been shown that the SCR system can remove dioxins from exhaust gases with an efficiency ranging from 90.5% to 97.4%. These results confirm that SCR technology, in addition to its primary function of removing NOx, can also be effective in reducing dioxins [60].

3.1.2. Pyrolysis

Pyrolysis has been used to produce charcoal from biomass for thousands of years. Historical evidence suggests that ancient civilizations used pyrolysis techniques to create charcoal, which was essential for various applications, including cooking, metallurgy, and as a heat source. For example, the ancient Egyptians used the liquid fraction obtained from the pyrolysis of cedar wood in their embalming process [61]. Interest in pyrolysis increased in the 1970s in response to the growing energy crisis [62–65]. Currently, there are about 80 suppliers of this technology worldwide. Most suppliers are located in Europe (42), followed by North America (21), with the remaining distributed in other regions, including China, Japan, and the Commonwealth of Independent States (CIS) [66]. A significant market segment is tire pyrolysis [67].

The pyrolysis process involves the thermochemical conversion of carbon-based chemical compounds in an oxygen-free atmosphere into simpler molecules. Since most of the reactions occurring during pyrolysis are endothermic, the conversion process requires the input of heat. The pyrolysis process can be expressed as:

$$C_aH_bO_c \overset{heat}{\rightarrow} \text{Solid} + \text{Liquid} + \text{Gas} + H_2O, \tag{2}$$

where Q is the heat that needs to be input to the reactor for the reactions to take place. The amount of heat Q that needs to be supplied to the system includes three components: the amount of energy required to evaporate water, the amount of energy necessary for the thermochemical conversion, and the amount of energy associated with covering losses due to radiation [68].

The main products of pyrolysis are three fractions with different states of matter: solid (known as char), liquid (known as pyrolysis oil), and gaseous (known as pyrolysis gas). Each fraction has its utility value:

1. Char, due to its high carbon content, has a higher heating value (HHV) ranging from 11 to even 23 MJ/kg, depending on the composition of the feedstock [69,70]. Char can contain up to 55% of the energy present in the original material [71]. The main factor reducing its calorific value is the ash content and other non-combustible parts.
2. Pyrolysis oil is a mixture of liquid hydrocarbons—including C8-C39 alkenes, C8-C20 alkanes, and aromatic compounds—as well as short-chain acids and alcohols [72]. Pyrolysis oil obtained from the conversion of polyolefins, with an HHV of up to 43 MJ/kg, can have properties similar to gasoline and diesel [73].
3. Pyrolysis gas is a mixture of gaseous hydrocarbons—including methane (CH_4), ethane (C_2H_6), and propane (C_3H_8)—hydrogen (H_2), carbon monoxide (CO), and carbon dioxide (CO_2) [74]. The gas has an HHV of about 18 MJ/kg and is easy to store and transport [74].

The proportion of each fraction in the final product depends on the substrate composition and process conditions. A typical thermogravimetric analysis (TGA) of organic substrates shows that organic matter degrades at temperatures from 200 °C to 700 °C, with degradation occurring in stages rather than linearly:

1. The first stage begins after 200 °C, with the degradation of simple organic acids and alcohols [75].

2. The second stage of decomposition, occurring between 220 °C and 400 °C, results from the decomposition of hemicellulose (220–315 °C) and cellulose (315–400 °C) [75,76].
3. The third stage of decomposition, occurring between 400 °C and 700 °C, results from the decomposition of lignin and other complex organic compounds, including plastics [25,75,76].

Most literature reports, covering a wide spectrum of waste materials, indicate that an increase in temperature above 600 °C is associated with an increase in the proportion of pyrolysis gas at the expense of the proportion of pyrolysis oil and char [61,68,77–80]. However, process temperature is not the only technological parameter. Alongside temperature, important technological parameters are the heating rate of the conversion zone and the conversion time. Based on this, pyrolysis can be divided into slow, fast, and ultra-fast:

1. Slow pyrolysis, also known as conventional pyrolysis, is dedicated to substrates with a particle size of <50 mm. The process involves slowly heating the biomass to the pyrolysis temperature at a rate of 0.1 to 1 K/s [81], followed by conversion at a temperature typically ranging from 400 to 500 °C for up to 2 h [81,82].
2. Fast pyrolysis is dedicated to substrates with a particle size of <3 mm. The reactor is heated at a rate of 10 to 200 K/s to a pyrolysis temperature typically ranging from 500 to 700 °C. The process requires a short residence time of vapors in the conversion zone (up to a few seconds) [81].
3. Ultra-fast pyrolysis, also known as flash pyrolysis, requires substrates with very fine granulation [81]. The reactor is heated at a rate exceeding 1000 K/s to a pyrolysis temperature above 700 °C. The residence time of vapors in the conversion zone is <0.5 s. In practice, laboratory studies using flash pyrolysis are conducted on waste with a particle size ranging from 125 µm to even 80 mm [74].

There are many literature reports on the efficiency of slow, fast, and ultra-fast pyrolysis. Studies indicate that conventional pyrolysis of waste up to 450 °C favors the formation of char, with its share exceeding 40% wt under these conditions, and above this temperature, the liquid fraction dominates, reaching up to about 60% wt. Fast pyrolysis at low temperatures allows for a high yield of char and liquid fraction, however, at 600 °C, there is a noticeable increase in the production of the gas fraction (even >75% wt). In studies on ultra-fast pyrolysis, waste with a high cellulose and lignocellulose content is more often used, allowing for the achievement of over 70% wt of the liquid fraction (up to 625 °C). Above 700 °C, the gas fraction dominates (Table 4).

Table 4. A summary of laboratory results on the pyrolysis of municipal waste or waste with properties similar to municipal waste.

HR	Temperature [°C]	Reactor Type	Feedstock | Size | Ash [%wt]	Share of the Fraction [%wt]			Ref.
				Gas	Liquid	Solid	
Slow pyrolysis	300	Horizontal tube	Lignin waste | <1 mm | N/A	8.93	18.13	72.94	[83]
	300	Horizontal tube	Food waste | N/A | 9.09	~15	~25	52.4	[84]
	400	Horizontal tube	Mixed MSW | <1 mm | 9.61	30.31	20.94	44.47	[85]
	450	Horizontal tube	Mixed MSW | <1 mm | 9.61	31.96	22.51	42.23	[85]
	500	Fixed-bed	Mixed MSW | <5 cm | N/A	8.92	61.26	29.82	[68]
	500	Fixed-bed	FWSD ** | 20–40 mm | N/A	5.3	52.2	42.5	[86]
	500	Fixed-bed	Food waste | 20–40 mm | N/A	7.4	60.3	32.3	[86]
	500	Horizontal tubular	RDF | <10 mm | 12.4	26	34	40	[87]
	500	Horizontal tubular	Biomass * | <0.2 mm | 14.18	21.99	55.12	22.89	[88]
	550	Horizontal tubular	Mixed MSW | <2 mm | 5.9	34	48	18	[82]
	600	Fixed-bed	Mixed MSW | <5 cm | N/A	13.41	59.98	26.61	[68]
	600	Horizontal tube	Food waste | N/A | 9.09	22.3	49.7	28	[84]
	650	Fixed-bed	Citrus residues | <850 µm | 2.85	<25.5	39.2	35.3	[89]
	700	Fixed-bed	Mixed MSW | <5 cm | N/A	13.41	59.98	26.61	[70]
	700	Horizontal tube	Lignin waste | <1 mm | N/A	28.09	32.12	39.79	[83]
	750	Horizontal tubular	Biomass * | <0.2 mm | 14.18	37.16	41.42	21.42	[86]
	800	Fixed-bed	Mixed MSW | <5 cm | N/A	31.16	52.61	16.23	[68]
	800	Fixed-bed	FWSD ** | 20–40 mm | 32.17	48.9	8.3	42.8	[86]
	800	Fixed-bed	Food waste | 20–40 mm | 16.96	67	8.2	24.8	[86]

Table 4. *Cont.*

HR	Temperature [°C]	Reactor Type	Feedstock ǀ Size ǀ Ash [%wt]	Share of the Fraction [%wt]			Ref.
				Gas	Liquid	Solid	
Fast pyrolysis	350	Horizontal tubular	Mixed paper/plastic ǀ 6 mm ǀ 10	9.4	6.2	12.8	[90]
	400	Fixed-bed	RDF ǀ 16/80 mm ǀ 17	18.6	30	49.8	[74]
	450	Fluidized-bed	MSW ǀ <1 cm ǀ 8.54	19	56.5	24.5	[91]
	500	Auger reactor	MGW *** ǀ 2 mm ǀ 0.93	27.5	31.2	41.3	[92]
	500	Horizontal tubular	RDF ǀ <0.2 mm ǀ 12.4	10	48	42	[87]
	550	Fluidized-bed	MSW ǀ <1 cm ǀ 8.54	10.1	38.4	21.5	[91]
	550	Horizontal tubular	Mixed paper/plastic ǀ 6 mm ǀ 10	14.9	40.2	29.4	[90]
	600	Fixed-bed	Mixed biomass ǀ <1 mm ǀ 1.02	33.53	47.94	18.53	[93]
	650	Fluidized-bed	Mixed MSW ǀ <1 cm ǀ 8.54	40.8	42.1	17.1	[91]
	700	Fixed-bed	RDF ǀ 16/80 mm ǀ 17	20.1	50	32.3	[74]
	750	Fixed-bed	Forestry residues ǀ 1.4–2.8 mm ǀ	60.13	20.11	14.37	[94]
	850	Fixed-bed	Forestry residues ǀ 1.4–2.8 mm ǀ	73.91	16.98	7.69	[94]
	950	Fixed-bed	Forestry residues ǀ 1.4–2.8 mm ǀ	79.27	12.21	5.44	[94]
	1000	Drop tube furnace	Woody biomass ǀ <0.8 mm ǀ 0.4	85	10	5	[95]
	1050	Fixed-bed	Forestry residues ǀ 1.4–2.8 mm ǀ	85.91	10.31	3.21	[94]
Flash pyrolysis	500	Horizontal tubular	Biomass * ǀ <0.2 mm ǀ 14.18	33.33	43.75	22.92	[88]
	550	Tubular transport	Rapeseed ǀ <1.8 mm ǀ 5.5	11	72	17	[96]
	600	Entrained-flow	Lignocellulosic ǀ <125 µm ǀ 3.87	5.37	73.74	20.89	[97]
	625	Ablative vortex	Pine ǀ <3.2 mm ǀ	13	71	18	[98]
	700	Horizontal tubular	RDF ǀ 16/80 mm ǀ 13	43.6	29	22.4	[72]
	750	Horizontal tubular	Biomass * ǀ <0.2 mm ǀ 14.18	55.53	25.96	18.51	[88]
	800	Horizontal tubular	RDF ǀ 16/80 mm ǀ 13	46.9	23	22.8	[72]
	850	Continuous flow	Light RDF foils ǀ 6/20 mm ǀ 23	55.2	12	32.8	[99]
	850	Continuous flow	Heavy RDF pack. ǀ 6/20 mm ǀ 23	62.6	5.8	31.4	[99]
	900	Horizontal tubular	RDF ǀ 16/80 mm ǀ 13	52.3	23	21	[72]
	950	Entrained-flow	Biomass ǀ <125 µm ǀ 1.2	68.5	16.8	12	[100]
	1000	Tubular	Wood ǀ 1–2 mm ǀ 0.5	>75	N/A	N/A	[101]

* Short-fiber waste from textile industry, ** food waste solid digestate, *** municipal green waste.

Depending on the type of substrates used in the pyrolysis process, the liquid phase can vary significantly in terms of chemical composition. As mentioned above, the liquid phase mainly consists of hydrocarbons, organic acids, and alcohols, but the exact proportions of these components depend on the feedstock material and technological conditions:

1. Organic and Food Waste Pyrolysis: Organic and food waste pyrolysis produces bio-oils with an even higher water content and a greater concentration of oxygenated compounds compared to woody or agricultural biomass. This high water and oxygen content leads to a lower energy density and reduced stability of the bio-oil. Oils derived from food waste often require further processing to remove water and reduce oxygen content to improve their energy value and stability. These oils are less suitable for direct use as fuels and need additional treatment, such as catalytic upgrading, to make them more usable as energy sources [102].
2. Plastic Wastes: The pyrolysis of plastic wastes results in liquids with high aromatic content, especially when the feedstock includes metals or catalysts. The composition of plastic-based liquid products depends on the proportion of paper, plastic, and metal in the feedstock. A high content of metals or catalytic additives increases the production of aromatic hydrocarbons, while paper-based substrates yield more water in the liquid fraction [103].
3. Agricultural Residues: The pyrolysis of agricultural residues, such as corncobs and straw, produces a two-phase liquid: an aqueous phase and an oil phase. The oil phase, rich in organic compounds, varies depending on the type of agricultural biomass used. Corncob pyrolysis yields a higher proportion of phenols and aromatics in the liquid phase, while straw produces more oxygenated compounds, affecting the fuel properties of the bio-oil [104].
4. Forestry Residues: Forestry residues, rich in extractives, produce a top-phase liquid fraction with high heating value, while the bottom phase resembles pyrolysis liquids from bark-free wood. The high extractive content in forestry residues leads

to phase separation in the liquid products, influencing their handling and storage properties [105].

5. Effect of Pyrolysis Conditions: The temperature and residence time during pyrolysis also impact the yield and quality of liquid products. For instance, polystyrene waste pyrolyzed at 450 °C produces a high yield of liquid oil with a high calorific value, consisting primarily of styrene, toluene, and ethyl-benzene [106].

The composition of pyrolytic oil based on the type of processed waste is summarized in Table 5.

Table 5. Composition of pyrolytic oil based on the type of feedstock.

Biomass Type	Key Liquid Products	Remarks	Ref.
Organic/Food waste	Water, organic acids, ox. compounds	High water content, low energy density	[102]
Plastic waste	Aromatics, water	Metal content increases aromatic fraction	[103]
Agricultural waste	Phenols, organic acids, aldehydes	Composition based on residue type	[104]
Forestry residues	Extractives, sugars, aldehydes	High extractive content leads to phase separation	[105]
Polystyrene waste	Styrene, toluene, ethyl-benzene	High liquid yield at 450 °C	[106]

3.1.3. Gasification

Gasification is a thermochemical process that involves converting a carbon-rich solid or liquid fuel into a gaseous fuel in the presence of a controlled amount of air or steam. The gasification process was initially dedicated to coal gasification. The first industrial-scale application of coal gasification took place in the early 19th century. William Murdoch, a Scottish engineer and inventor, was the first to use coal gas to light his house in Redruth, Cornwall, in 1802. This event marked the beginning of gas lighting technology and led to the establishment of the first gasification plants in England. In 1812, the Gas Light and Coke Company was founded in London, which began large-scale gas production, supplying it for street lighting and private homes. This event is considered the beginning of the commercial use of coal gasification [107]. A significant technological advancement was the development of the fluidized bed gasification process by engineer Karl Winkler in 1922 [108]. This process, commercialized by Rheinische Braunkohlenwerke AG, allowed for more efficient and uniform conversion of coal into synthesis gas. In 1936–1937, fixed-bed reactors developed by Lurgi AG began to be used in Germany. The so-called Lurgi process was developed mainly for the production of synthesis gas used for chemical synthesis and as fuel gas [109].

Gasification is a complex thermochemical process that involves several key stages enabling efficient feedstock conversion. These stages include drying, pyrolysis, oxidation (combustion), and reduction. A typical gasification process is schematically presented in Figure 2.

The first stage of gasification, drying, involves the evaporation of water contained in the feedstock at temperatures below 150 °C. Water vapor plays a key role in subsequent stages, enabling reactions such as the water–gas shift reaction and steam reforming.

After drying, the feedstock undergoes pyrolysis, where it is heated in an oxygen-free environment to temperatures between 200 and 700 °C. In this process, organic materials decompose into three fractions: gas, liquid, and solid. The formation mechanisms of these fractions are detailed in Section 3.1.2 Pyrolysis. The carbon from the solid fraction participates in later thermochemical reactions.

The next stage is oxidation, occurring at temperatures from 700 to 1200 °C, where the feedstock reacts with a controlled amount of oxygen or air. Oxidation reactions are fundamental in gasification, providing the energy needed to maintain high temperatures in the reactor.

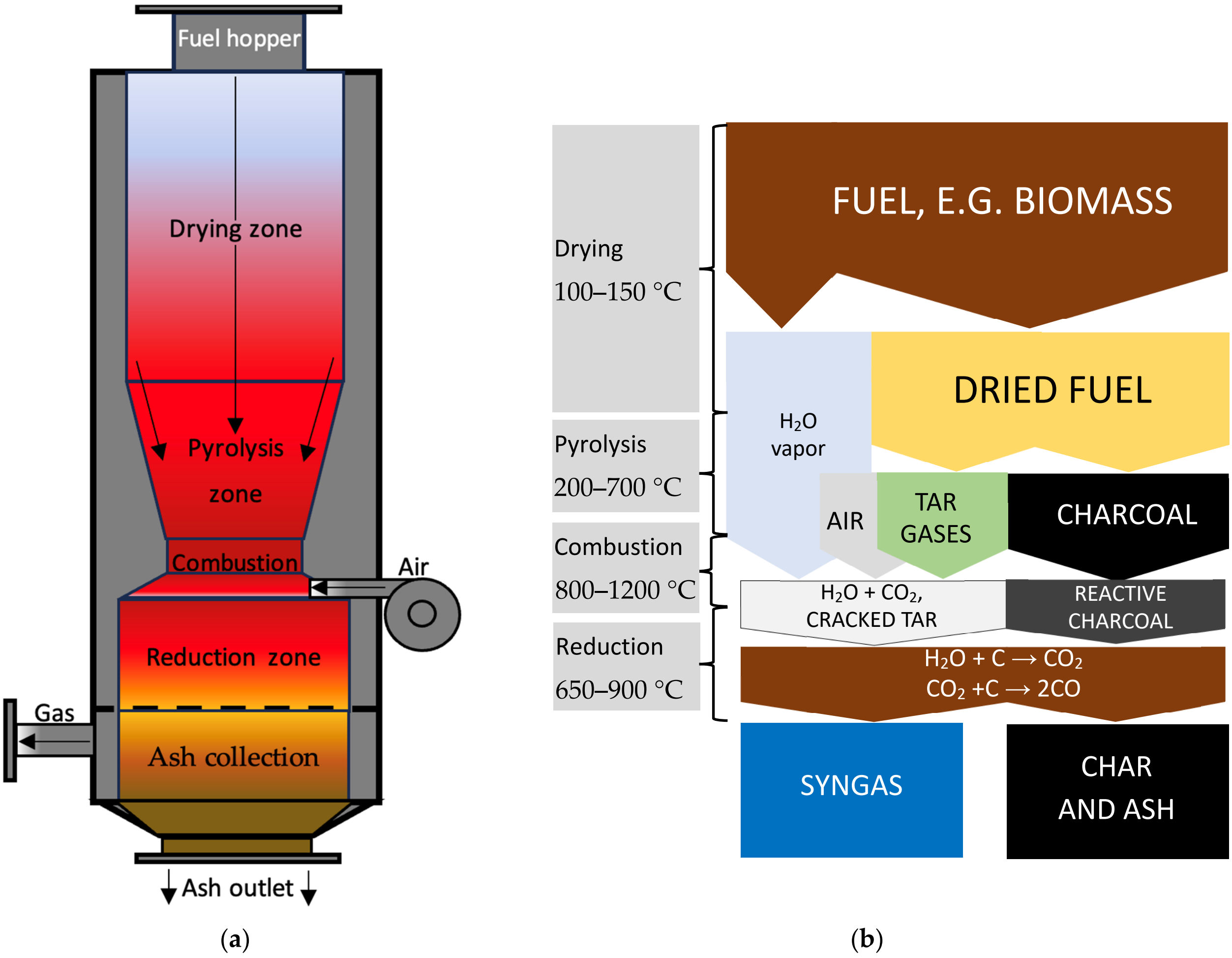

Figure 2. Conceptual diagram of the waste gasification process; (**a**) typical fixed-bed downdraft gasifier (source: based on [110]); (**b**) stages of the gasification process (source: based on [111]).

The final stage is reduction, taking place in the lower part of the reactor. Here, the gases from oxidation (CO and CO_2) react with charcoal at temperatures between 800 and 1600 °C, producing hydrogen (H_2) and additional carbon monoxide (CO). These reactions are crucial for producing highly calorific syngas, with typical reactions in this zone including the water–gas reaction and the Boudouard reaction.

Understanding the thermochemical reactions during waste gasification is essential for comprehending the conversion of waste into valuable syngas. These reactions include both exothermic processes, which release heat, and endothermic processes, which require energy input. The most important reactions are presented in Table 6.

Table 6. The main thermochemical reactions occurring during waste gasification (source: [112–119]).

Type of Reaction	Reaction	Enthalpy [MJ/kmol]
Oxidation reactions:		
Carbon partial oxidation	$C + ½ O_2 \rightarrow CO$	−111
Carbon monoxide oxidation	$CO + ½ O_2 \rightarrow CO_2$	−283
Carbon oxidation	$C + O_2 \rightarrow CO_2$	−394
Hydrogen oxidation	$H_2 + ½ O_2 \rightarrow H_2O$	−242

Table 6. *Cont.*

Type of Reaction	Reaction	Enthalpy [MJ/kmol]
Gasification reactions involving water vapor:		
Water–gas reaction	$C + H_2O \rightarrow CO + H_2$	+131
Water–gas shift reaction	$CO + H_2O \rightarrow CO_2 + H_2$	−41
Steam methane reforming (I)	$CH_4 + H_2O \rightarrow CO + 3\,H_2$	+206
Steam methane reforming (II)	$CH_4 + 2\,H_2O \rightarrow 2\,CO_2 + 4\,H_2$	+165
Steam reforming	C_nH_m + n H2O → n CO + (n + m/2) H_2	Endothermic
Gasification reactions involving hydrogen:		
Hydrogasification	$C + 2\,H_2 \rightarrow CH_4$	−75
Methanation	$CO + 3\,H_2 \rightarrow CH_4 + H_2O$	−227
Gasification reactions involving carbon dioxide:		
Boudouard reaction	$C + CO_2 \rightarrow 2\,CO$	+172
Dry reforming	$C_nH_m + n\,CO_2 \rightarrow 2n\,CO + m/2\,H_2$	Endothermic
Gasification reactions involving hydrocarbons:		
Ethylene formation	$2CH_4 + C_2H_4 \rightarrow CO + 2\,H_2$	+202
Gasification reactions involving ammonia:		
Ammonia formation	$3\,H_2 + N_2 \rightarrow CO + 2\,NH_3$	+92

Based on the above analysis, it can be concluded that the oxidation stage is essentially the only exothermic stage during the waste gasification process. This naturally raises the question regarding the net energy gain (positive or negative) of the entire process. The energy balance of the gasification process can be most simply represented as:

$$\Delta Q = Q_{rec} - Q_{in} - Q_{rad} \tag{3}$$

where Q_{rec} is the energy recovered in the form of syngas, Q_{in} is the total energy input to the system, including the chemical energy of the waste, the energy needed for drying, and the energy required for the gasification process itself, Q_{rad} includes losses due to radiation and other energy losses within the system. Thus:

$$Q_{rec} = \mathrm{HHV} \cdot m_f \cdot \eta \tag{4}$$

and

$$Q_{in} = Q_{ev} + Q_{en} \tag{5}$$

where HHV is the higher heating value of the feedstock [J/g], m_f is the mass of the feedstock [g DM], η is the energy efficiency of the gasification process, Q_{ev} is the energy needed to evaporate the water, and Q_{en} is the endothermicity of the entire process. Ultimately:

$$Q_{ev} = m_w \cdot (4,18\mathrm{J/g^{\circ}C} \cdot \Delta T + 2260\mathrm{J/g}) \tag{6}$$

where 4.18 J/g°C is the specific heat capacity of water, ΔT is the change in temperature, and 2260 J/g is the heat of vaporization of water.

The above model can be used for a preliminary assessment of the energy profitability of gasifying a specific type of substrate. It is important to note that the gasification process uses substrates such as biomass, municipal solid waste (MSW), high-calorific plastic wastes, agricultural wastes, and sewage sludge [120–125]. In Table 6, calculations for the gasification of two types of municipal waste weighing 1 kg each are presented under standard laboratory conditions (standard ambient temperature and pressure (SATP)), i.e., 25 °C and 1 atm, assuming an η of 31% [126], a Q_{en} of 3,05 kJ/g [119], and Q_{rad} of 10% of Q_{in} (Table 7.).

Table 7. Theoretical energy balance for the gasification of two different samples of municipal waste weighing 1 kg, conducted under SATP conditions.

Type of Waste	HHV [MJ/kg]	Humidity [%wt]	Feedstock		a	b	c = a + b	d	e	e-c-d
			m_f [kg]	m_w [kg]	Q_{ev} [MJ/kg]	Q_{en} [MJ/kg]	Q_{in} [MJ/kg]	Q_{rad} [MJ/kg]	Q_{rec} [MJ/kg]	ΔQ [MJ/kg]
Mixed MSW	15	40	0.6	0.4	1.03	3.05	4.08	0.41	2.79	−1.70
RDF	22	12	0.88	0.12	0.31	3.05	3.36	0.34	6.00	2.31

In the above example, the energy balance of gasification of mixed municipal waste is negative and amounts to −1.70 MJ/kg, while the gasification of RDF is positive and amounts to +2.31 MJ/kg. The selection of the substrate is, of course, not the only factor affecting the profitability of the process. The profitability of the entire investment also depends on logistical costs and the current costs of waste disposal in the relevant area.

The above example also does not include the application of heat recovery from a cogeneration engine and other process improvements. In studies of [127], the application of heat recovery during microcogeneration of biomass increased thermal efficiency from 11.3% to 56.2%. It is important to note that, theoretically, the energy efficiency of the biomass gasification process can reach even 81.5% [128], so it is necessary to search for other solutions and technical and technological improvements. For example, using syngas in reciprocating engines, systems have demonstrated a cold gas efficiency of 57–60% and overall electrical efficiency of about 22.5%. Efficient heat recovery can significantly boost overall thermal efficiency [129]. Advanced systems combining solid oxide fuel cells with gas turbines and Kalina cycles have shown energy efficiencies up to 47.49% and exergy efficiencies up to 36.14% [130]. The use of plasma gasification of refuse-derived fuel can achieve a cold gas efficiency up to 74.8%, making it a competitive option compared to traditional methods [131].

4. Current Research Trends in Thermochemical Conversion Technologies

4.1. Incineration

One of the current research directions is the implementation of municipal waste incineration (MSWI) technology in circulating fluidized-bed (CFB) systems [132]. The high efficiency and environmental benefits of CFB technology have been known for years [133,134]. However, this technology operates at relatively low temperatures and may require appropriate fuel preparation [135]. Nonetheless, progress has been made in this area in recent years. Recently, the technology of incineration in an internal circulating fluidized-bed (ICFB) boiler has been developed, which increases the efficiency of waste material incineration compared to the conventional CFB method [136]. ICFB systems are characterized by fast combustion reactions, low emissions, and uniform bed temperature. Experiments conducted on prototype ICFB systems have shown that this technology is effective in incinerating low-grade fuels, such as municipal waste [137]. An example of an ICFB installation implemented under technical conditions is the installation at Jeonju Paper Corporation in Korea, which generates 33 MW of electric power from the incineration of wood waste, which is the maximum efficiency of this technology [136]. The installation was developed by Kawasaki [136].

The development of small-scale waste incinerators (2 Mg/h [138]) remains a current research direction. The main obstacles to the development of this technology include low energy efficiency. Recent progress has been observed in technological development. Optimization of air injection into the primary combustion chamber (PCC) and secondary combustion chamber (SCC), recirculation of hot flue gases, and redesign of the feed system allowed [139] the design of compact-sized installations operating in a batch system (1 Mg of waste per cycle every 6–8 h). The development of small-scale MSWI is greatly influenced by the use of the organic Rankine cycle (ORC). The ORC technology operates on a principle similar to the conventional Rankine cycle, but instead of water, it uses organic working fluids with lower boiling points. This allows for more efficient conversion of low-temperature waste heat into electricity. According to [140], the use of the ORC increases the energy efficiency of municipal waste incineration by about 60%. The most commonly used organic working fluids include R245fa (1,1,1,3,3-pentafluoropropane), toluene, R123 (dichlorotrifluoroethane), methanol, ethanol, R134a (tetrafluoroethane), pentane, and R227ea (heptafluoropropane) [141–146]. Each of these compounds has its drawbacks, requiring additional precautions—for example, R245fa is a material with a high GWP index, while toluene, methanol, and ethanol are flammable liquids.

An interesting research direction is the miniaturization of waste incineration installations for domestic use. Current designs of domestic incinerators show significant efficiency, achieving combustion temperatures above 1000 °C and reducing waste volume by up to 90% [147]. An example of a technological solution is the use of a fluidized reactor powered by HHO gas (a mixture of hydrogen (H_2) and oxygen (O_2) in a 2:1 ratio). A typical method of producing HHO gas is water electrolysis, which—according to [148,149]—can be achieved under domestic conditions using photovoltaic panels.

4.2. Pyrolysis

Pyrolysis of MSW is gaining popularity as an effective waste management method that allows for energy recovery and waste volume reduction. Over the past decade, research on MSW pyrolysis has significantly advanced, focusing on process optimization and the integration of pyrolysis with other solutions. Separating the drying chamber from the pyrolysis chamber and recirculating waste heat to the combustion chamber enabled [150] the pyrolysis of waste with up to 60% moisture content, achieving an energy utilization rate of up to 80% without the need for additional energy sources. New design solutions include the use of microwave radiation [151,152]. The microwave pyrolysis process involves transferring microwave energy to polar materials, which absorb it and convert it into thermal energy needed to reach pyrolysis temperatures. However, an effective microwave pyrolysis process requires the addition of carbon/activated carbon to the substrate as a material that supports microwave absorption [151,152]. Current research focuses on optimizing the parameters of the microwave pyrolysis process, such as microwave power, to maximize process efficiency. According to [153], microwave pyrolysis provides faster and more uniform heating of the material compared to traditional pyrolysis, resulting in lower energy consumption. Moreover, the products obtained from pyrolysis typically have a higher calorific value.

In recent years, the efficiency and quality of pyrolysis end products have been significantly improved by the implementation of catalytic substances and materials. The impact of catalysts on the pyrolysis process includes changes in the chemical composition of pyrolysis products, increased process efficiency, and reduced emissions of harmful substances. Different types of catalysts, such as zeolites, metal oxides, and composites, exhibit varying catalytic properties:

1. Zeolites, such as ZSM-5, Y-zeolite, and β-zeolite, enhance the yield of volatile fractions and accelerate the breakdown of polymer chains, converting aliphatic hydrocarbons into aromatic and cyclic compounds in pyrolytic oils [154]. In the study of [155], Y-zeolite and ZSM-5 were used in amounts ranging from 0.1 to 2.0 wt% of the feedstock. Increasing the Si/Al ratio in the catalyst influenced the increase in gas fraction production and the reduction of the molecular weight of liquid fraction products.
2. Metal oxides, such as MgO, exhibit high deoxygenation activity, which improves the quality of pyrolytic oils. The study of [156] demonstrated that the use of MgO on activated carbon (AC) in doses ranging from 5 to 30 wt% of the reactor feed resulted in a decrease in liquid fraction yield and an increase in gas and solid fraction yields.
3. Iron-based catalysts reduce activation energy, resulting in higher production of combustible gases and a reduced amount of oxygen-containing compounds in pyrolytic tars. The study of [157] investigated the impact of iron-based additives (iron ore and

iron oxide) on municipal waste pyrolysis; catalyst doses ranged from 10 to 30 wt% of the feed. The addition of iron reduced the activation energy of the process and increased the efficiency of MSW conversion to 55.81%.

4. Composites, such as Ni-Mo/Al_2O_3 and Ni/ZSM-5, allow for increased hydrogen content in the gas fraction and aromatization of the liquid fraction. In a study [158], high concentrations of Ni/ZSM-5 and Ni/SAPO-11 composites also increased the production yield of both fractions.
5. The study of [159] showed that using calcined dolomite as a catalyst at a feed-to-dolomite weight ratio of 5:1 wt significantly increased the gas fraction production yield compared to the process without a catalyst. Calcium oxide, CaO, promotes secondary cracking and reduction reactions, which in turn increases the production of the gas fraction [160].

From the above review, it is evident that catalytic compounds of pyrolysis constitute a relatively high share of total feedstock, which may lead to increased operational costs. Hence, the use of low-cost natural catalysts, such as natural zeolite, bentonite, and olivine [161], as well as waste-derived catalysts, such as oyster shells [162], is recommended. In analyses [163], conducted in 2016, the production cost of pyrolytic oil from biomass using ZSM-5 catalyst was estimated at USD 615/MT (USD 22/GJ), with potential fluctuations ranging from USD 615 to USD 841/MT.

4.3. Gasification and Other Thermochemical Conversion Technologies

In recent years, the efficiency and quality of gasification products have been significantly improved by the implementation of catalysts. The impact of catalysts on the gasification process includes changes in the composition of gas products, increased process efficiency, and reduced emissions of harmful substances. Different types of catalysts, such as dolomite, metal oxides, and composites, exhibit varying catalytic properties.

1. Dolomite: It is widely used as a catalyst in gasification processes, especially for biomass substrates. Dolomite effectively reduces tar content and promotes carbon conversion into syngas. Studies have shown that dolomite significantly improves gas efficiency, particularly in relation to hemicellulose, while its impact on other biomass components, such as cellulose and lignin, is more limited [164].
2. Metal Oxides such as Ni and Fe: Metallic catalysts, such as nickel (Ni) and iron (Fe), are widely used in steam-reforming processes and tar reduction. In gasification processes, Ni/Fe acts as a dual-function catalyst, allowing for more efficient hydrogen production and increased resistance to carbon deposition. For example, nickel and metal-oxide-based catalysts have demonstrated the ability to improve methane r-forming efficiency at moderate temperatures (400–600 °C), reducing CO and CO_2 content [165].
3. Composite Catalysts, such as Rh/CeO_2: High-performance composite catalysts, such as Rh/CeO_2, are used in biomass gasification at lower temperatures, leading to higher carbon conversion and minimal tar formation. The use of such catalysts significantly increases process efficiency, even at lower temperatures compared to conventional methods [166].
4. Alkaline Catalysts: Alkaline metal salts, such as Na_2CO_3, exhibit high activity in the gasification process, particularly with carbon. These catalysts can significantly lower the activation energy and increase carbon conversion in syngas. Na_2CO_3 has proven to be highly effective during biomass and coal gasification, contributing to the reduction of tar content [167].

In the context of municipal waste gasification, the technology worth mentioning is plasma temperature gasification. Plasma temperature in the reactor is controlled using an electric arc. The electric arc is generated between two electrodes in a specially designed plasma torch. These electrodes are usually made of materials resistant to high temperatures, such as graphite or tungsten. Low-voltage plasma torches utilize voltages in the range of

100–300 V and currents of several hundred amperes. These torches are used for lower-power applications, such as laboratory research or small installations. High-voltage plasma torches utilize voltages in the range of 1000–2000 V and currents from 1000 to 10,000 A. These are high-power systems used in industrial plasma reactors for large-scale waste gasification. High voltage causes the electric current to flow between the electrodes, resulting in the ionization of the gas (plasma generation) around the electrodes. The gas flow through the electric arc is usually forced, with the oxidizing gases typically being air or steam. A plasma torch generates very high temperatures, typical temperatures for air gasification processes being around 800–1300 °C, and for steam gasification—1000–1400 °C [168]. In designing reactors for plasma waste gasification, a key technological aspect is ensuring homogeneous heating and mixing of the material, allowing for its uniform conversion into syngas [169,170]. The high entry barrier, including high investment costs, means that the number of operational plasma gasification installations on a technical scale is relatively small, and the technology is targeted at a narrow market segment—plasma gasification is used, for instance, for the disposal of municipal waste generated on cruise ships and military aircraft carriers [171].

Modeling of municipal waste gasification at different plasma temperatures (1500, 2000, 2500 °C) showed that, at 2000 °C, it is possible to obtain syngas with a calorific value of 6.02 MJ/Nm^3, with a gasification efficiency of 49.2% [172]. This is roughly consistent with the results obtained by [131], who, in their studies on RDF gasification in a single-phase plasma reactor, achieved a cold gas efficiency of 56%, with high levels of CO and H_2 and low tar content. For comparison, the efficiency of coal conversion in the plasma gasification process using plasma ranges from 80% to 100%, depending on the reacting gases and operating conditions [131]. In studies on the gasification of glycerol using water plasma, the energy conversion efficiency exceeded 63.86%wt and the LLV of the syngas exceeded 9.82 MJ/Nm^3 [173].

A promising technology for the thermochemical processing of municipal solid waste is hydrothermal treatment of municipal solid waste (HTMSW). HTMSW involves processing waste placed in a closed reactor using high-temperature steam under high pressure. Typical conditions include temperatures ranging from 180 °C to 235 °C, pressures around 2 MPa, and treatment times from 30 to 90 min [174,175]. Hydrothermal treatment increases the energy density of processed municipal waste, making it comparable to subbituminous coal, and the calorific value of processed waste can reach up to 24 MJ/kg [172]. The treated waste exhibits better combustion properties and reduced emissions of sulfur dioxide (SO_2) and nitrogen oxides (NO_x) during cofiring with coal [176,177]. Hydrothermal treatment significantly reduces the chlorine content in municipal waste, which is crucial for minimizing corrosion and dioxin formation in incinerators. Organic chlorine compounds are converted into water-soluble inorganic compounds, which can be washed out, reducing chlorine content to about 0.2% [176]. Hydrothermally processed municipal waste can be cofired with coal in existing power plants, reducing coal demand and improving the efficiency of energy recovery processes [175]. A commercial-scale hydrothermal plant in Indonesia demonstrated the feasibility of processing 50 tons of municipal waste per day, producing solid fuel that can be used for energy production or cement manufacturing [178].

4.4. Summary of the Section

The literature review conducted in this section clearly indicates ongoing progress in the development of thermochemical waste conversion technologies. Such an approach not only contributes to improving energy efficiency but also supports the potential use of municipal waste as a resource.

5. Environmental Impacts

The impact of municipal waste incineration on air quality has been a topic of intense debate and research for decades. This section delves into the intricate relationship between waste management practices and the emission of hazardous air pollutants, offering a comprehensive overview of how our understanding and control of these emissions have evolved over time. From the discovery of toxic dioxins to the implementation of advanced emission reduction technologies, this section highlights the significant strides made in mitigating environmental and health risks.

5.1. Air Quality and Emissions

For many years, the incineration of municipal waste was seen as an effective way to reduce its volume and generate energy. In 1971, Dr. James R. Arnold from the University of California in San Diego [179] first demonstrated the toxic nature of dioxins. The link between the production of PCDD and the process of municipal waste incineration led to further research on the emission of toxic compounds during municipal waste incineration. In the 1980s [180], in addition to PCDD, harmful air pollutants (HAPs) emitted by incinerators included: polychlorinated dibenzofurans (PCDFs), heavy metals, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), chlorobenzenes, chlorophenols, halogenated organic acids, phthalates, aldehydes, ketones, alkenes, and acids (e.g., HCl). At the same time, the low efficiency of HAP purification systems in existing installations was emphasized [180].

A great deal of research was conducted on this subject. Among the potential hazards caused by HAPs, respiratory diseases, cancers, reproductive problems, increased risk of congenital defects, and endocrine disorders in children were mentioned [181]. Research carried out by [182] in the early 1980s demonstrated that high air pollution levels in MSWI areas had a negative impact on lung function in children. Studies in Japan, Spain, and Germany revealed that incinerator workers and residents living nearby have significantly higher levels of dioxins, furans, and PCBs in their blood or urine [183–192]. Epidemiological analyses around 52 MSW incinerators in the UK confirmed the existence of an excessive risk of primary liver cancer in the period 1974–1986 within a distance of <1 km from the incinerators [193]. In Italy, a more than three-fold higher risk of lung cancer was demonstrated for the periods 1979–1981 and 1985–1986 [194]. In France, a link was shown between dioxin emissions (16.3 ng TEQ/m^3) and lymphoma incidence in the period 1980–1995 within a radius of 3 km from an incinerator [195]. Studies conducted in the 1990s by [196] indicated the possibility of HAP bioaccumulation in the food chain, posing an indirect threat to human health.

The above publications sparked public debates and social movements. An example of social protests against the construction of waste incinerators in the 1990s is the protests of the residents of Pilsen in the Czech Republic. In 1996, Pilsen residents protested against plans to build a waste incinerator, fearing negative impacts on the environment and public health. These protests were part of a broader movement against the construction of new waste incineration installations in Central and Eastern Europe after the fall of communism [197].

Since around 2000, the situation regarding emissions of pollutants into the environment has been systematically improving. This was the result of implementing increasingly effective methods for reducing HAP emissions, such as bag filters, scrubbers, and electrostatic precipitators, which effectively reduce pollutants like dioxins and furans [47]. Safety in this area also increased due to innovations in high-temperature incineration and rapid cooling of flue gases, which minimized the risk of dioxin reformation [55]. Studies conducted from 2003–2004 still reported an excess of respiratory diseases or specific cancers in the incinerator area, but no convincing evidence of a link between incinerators and cancers was found [198,199]. The issue was taken very seriously. In Portugal, extensive biomonitoring programs were implemented to assess the impact of waste incinerators on human health [200]. These programs included several specific analyses, such as blood analysis (including umbilical cord blood) and breast milk analysis, for levels of dioxins and heavy metals. This biomonitoring did not show significant differences in levels of dioxins and heavy metals compared to control populations, indicating the high effectiveness of modern emission reduction technologies compared to older technologies. In the UK, studies were conducted on the spatial variation of PM10 concentrations and their impact on public health in the period 2003–2010 [201]. In the UK, from 2003 to 2010, studies were conducted on PM10 emissions from 22 municipal waste incinerators. The analysis focused on assessing the spatial variation of PM10 concentrations and their impact on public health. The results showed minimal impact on local PM10 concentration levels, which constituted a small fraction of the background concentrations, ranging from 6.59 to 26.8 µg/m. Nevertheless, the report indicated short-term exceedances of PM10 emissions, which were associated with installation failures. Similar results were shown by studies in Korea [202], although the authors considered the impact of HAPs on the population to be "substantial" due to the high population density in Seoul.

From 2003–2010, a detailed analysis of harmful air pollutant emissions was conducted in China [203]. The research concerned both the type and load of HAPs emitted by incinerators throughout the country. The studies showed a successive, almost linear increase in HAP emissions during the study period, especially in the eastern and southeastern regions of the country, characterized by rapid economic development and high population density. For example, nitrogen oxide (NO_x) emissions increased from 4393.6 Mg in 2003 to 28,471.1 Mg in 2010, sulfur oxide (SO_x) emissions from 1694.3 to 12,062.1 Mg, and PCDD/Fs emissions increased from 2.5 to 23.6 g TEQ. Nevertheless, it should be noted that, since 2001, the production of municipal waste in China has increased from 135 million Mg to even 400 million Mg, and the capacity of waste incinerators has increased from 2.75 million tons to over 100 million Mg/a since 2002 (sic) [204]. Therefore, we are talking about a completely different scale of waste management compared to the rest of the globe. Moreover, to estimate the real impact of waste incinerators on the level of dioxins in the environment, it would be necessary to determine the level of emissions from other industrial and non-industrial sources. For example, according to the UK Environment Agency (DEFRA), 15 min "fireworks" set off to celebrate the new millennium in London in 2000 (about 35 Mg of pyrotechnic materials) resulted in the production of more dioxins than 120 years of operation of a waste-to-energy plant (municipal waste incinerator) in southeastern London (SELCHP—capacity 420,000 Mg/year). However, some opinions, such as that of UKWIN, question the scientific nature of this information [205].

In recent years, the regulations for hazardous air pollutant emissions have become more rigorous. This practice is global and constitutes an additional factor in the development of technological innovations, including in the field of emission reduction and monitoring of HAP emissions. Global emission standards are similar, although the most stringent regulations in this regard seem to apply in the United States. Interestingly, in the areas analyzed, there are no specific standards for CO_2 emissions from waste incinerators. In the European Union, unlike pollutants such as NOx, PM, or dioxins, CO_2 is not classified as a substance subject to direct restrictions in these facilities. Instead, CO_2 emissions are regulated under the broader EU Emissions Trading System (ETS), which imposes caps on greenhouse gas emissions at a market level, rather than on individual facilities. Similarly, in the United States, waste incinerators are not subject to direct CO_2 emission limits but must comply with regional or federal greenhouse gas policies, such as cap-and-trade systems, which aim to reduce emissions across various sectors. In China and Japan, CO_2 emissions from waste incinerators are also covered by national climate policies; however, like in the EU and the US, there are no specific emission limits for CO_2 from these facilities. The policies of these countries focus on broader targets for reducing greenhouse gas emissions as part of their climate strategies.

Although the current norms and standards do not regulate the permissible CO_2 concentrations in exhaust gases, it is justified to limit the emission of this gas. This is due to the need to combat global warming and results from the environmental policies of many countries. Therefore, not only low-emission and emission-free technologies are being sought but also solutions for the capture, separation, transport, and long-term storage of CO_2. Methods of carbon capture and utilization (CCU) include mineral carbonization [206], sequestration in geological formations and physical processes (adsorption, membrane separation) [207], the cryogenic CO_2 capture system [208], chemical processes (chemical adsorption and absorption) [209], and biological processes (afforestation, ocean fertilization, and use of photosynthetic microorganisms) [210]. One promising solution is the use of fast-growing microalgae, which are able to effectively bind CO_2 and assimilate other gaseous pollutants (NO_x, SO_x) during photosynthesis [211]. Microalgae biomass can be used for energy purposes [212] or for the production of valuable substances with high economic potential, including lipids [213], fatty acids [214], dyes [https://doi.org/10.3390/app14031104, accessed on 20 August 2024], fertilizers [215], and others.

The permissible emission standards for basic pollutants from the incineration of municipal waste in selected countries are listed in the Table 8.

Table 8. Emission limits of pollutants generated during municipal waste incineration in various regions of the world. For comparison purposes, the data were converted to [mg/m^3]. Countries are presented in alphabetical order (source: [216]).

HAP	Unit	China	Japan *	South Korea	EU	USA **
PM	mg/m^3	20	24	20	10	25 | 7.4
NO_x	mg/m^3	250	307.5	143.5	200	180 | 110
SO_x	mg/m^3	80	85.8	85.8	50	29 | 20
HCl	mg/m^3	50	41	32.8	10	29 | 13
CO	mg/m^3	8	N/A	62.5	50	<125
Cd	mg/m^3	0.1	N/A	0.05	0.05	0.035 | 0.0015
Hg	mg/m^3	0.05	0.08	0.05	0.05	0.05 | 0.012
Pb	mg/m^3	1.6	0.2	N/A	0.5	0.4 | 0.056
PCDD/Fs	ng I-TEQ/m^3	0.1	0.1	0.1	0.1	0.1

* Limits for MSWC of > 250 Mg/d that begin construction or undergo substantial modification; ** current limits | proposed limits for MSWC of > 250 Mg/d.

Modern waste incineration plants, equipped with advanced pollution control technologies, effectively meet current, stringent emission requirements and do not pose significant cancer risk or negative impact on reproduction or development [217]. An analysis of municipal waste incinerators in China, conducted in 2017, showed a significant improvement in air pollution control systems and compliance with new emission standards [218]. Long-term biomonitoring around Dutch waste incinerators indicated that the concentrations of heavy metals, PAHs, and dioxins/PCBs were similar to background levels and did not exceed maximum allowable concentrations [219]. A comparative study of emissions from modern municipal waste incinerators and typical heating systems in Poland showed that waste incinerators have lower pollutant emissions than traditional coal-fired boilers [220]. HAP emission standards are also met in cement kilns cofiring municipal waste [215,221].

Discussing emission standards for pyrolysis and gasification is more complex. The classification of these thermochemical processes as either "incineration" or "chemical recycling" remains contentious. For example, the European Parliament Directive on industrial emissions [222] classifies pyrolysis and gasification installations as "waste incineration plants". However, according to Article 42 of this Directive, the provisions shall not apply to gasification or pyrolysis plants if the gases resulting from this thermal treatment of waste are purified to such an extent that they are no longer a waste prior to their incineration and they can cause emissions no higher than those resulting from the burning of natural gas. On the other hand, the EPA proposes to revise the definition of "municipal waste combustion unit" in 40 CFR 60.2977 and 40 CFR 60.3078 of the Code of Federal Regulations of the United States by removing the reference to "pyrolysis units" from the definition, "reflecting our view that such units should not be regarded as municipal waste combustion units" [223]. These actions are inconsistent with EU regulations, even in the context of the very definition of waste recycling, which—according to Directive 2008/98/EC on waste—"does not include energy recovery and the reprocessing into materials that are to be used as fuels or for backfilling operations" [224].

Nevertheless, research on emissions from pyrolysis and gasification processes is being conducted. The availability of detailed results is relatively lower than those concerning direct incineration due to the marginal share of pyrolysis and gasification installations processing municipal waste under technical conditions [225]. Generally, compared to pyrolysis, gasification generates smaller amounts of pollutants, particularly in terms of particulate matter (PM) and PCDD emissions. In the gasification process, there is the possibility of recycling condensing gases, which further reduces the level of HAP emissions. However, since pyrolysis occurs at lower temperatures and without the presence of oxygen, NOx and SOx emissions are higher during gasification.

PM emissions typically range from a few milligrams to hundreds of milligrams per cubic meter of exhaust gas [226]. Studies on pyrolysis and gasification of biomass have shown that particulate matter (PM10) emissions are higher during pyrolysis, at temperatures of 500–700 °C, with particles mainly in the size range of 0.25–2.5 µm [227].

Studies [228] have shown that in gasification systems, especially in three-stage gasification systems, process conditions (particularly high temperature) allow maintaining dioxin (PCDD/F) emissions at levels from 0.068 to 0.085 ng I-TEQ/Nm^3, which are significantly lower compared to pyrolysis. Research findings [229] on pollutant emissions during pyrolysis and combustion of polyurethane foam at different temperatures showed variation in PCDD emissions depending on temperature and oxygen presence. The production of PCDD/F during pyrolysis increased up to 750 °C and then sharply decreased at 850 °C, suggesting their thermal degradation. The highest PCDD/F emission was obtained during waste combustion at 850 °C (414.5 ± 21 pg/g of sample).

Emissions of nitrogen oxides (NO_x) from pyrolysis plants typically range from 1.1138 mg/Nm^3 to 131 mg/Nm^3 [230]. Due to the process specificity, NO_x emission levels during pyrolysis appear to be lower than during gasification. Studies on the pyrolysis of various solid wastes conducted by [231] showed NO_x emissions ranging from 0 to 20 ppm. Analysis of emissions from pyrolysis products showed that burning pyrolytic bio-oil generates NO_x emissions at 16.8 ppm, which complies with EPA standards for waste incinerators [232]. Studies on the pyrolysis of furniture waste and tires showed that NOx emissions were low, even with high nitrogen content in the waste [233]. Research on the control of NO_x precursors during sewage sludge pyrolysis indicated that adding red mud can reduce NO_x precursor emissions (NH_3 and HCN) by 15.10% for NH_3 and 24.72% for HCN at 900 °C [234]. Studies on the gasification of organic waste showed that NO_x emission was maintained at a maximum level of 76.66 ppm [235]. In experiments using an updraft gasifier, the NOx concentration was a maximum of 77 ppm [236]. Studies on gasification of solid waste in Brazil showed that NOx concentration was lower than the values established by Brazilian regulations, at 67 ppm [237]. Studies on gasification and combustion of municipal waste indicated that NO_x concentration was around 50 ppm, which is significantly lower than during waste combustion [238]. Emission levels in studies [239] on biomass gasification, depending on the type of feedstock used, showed that NO_x concentration ranged from 80 to 150 ppm for wood biomass and from 450 to 900 ppm for corn grain gasification. NH_3 concentration was respectively from 200 to 1100 ppm (wood biomass) and 8900 ppm (corn grain). When these values are converted to mg/m^3 for the conditions specified in the IED Directive [240], i.e., at a temperature of 273.15 K, under a pressure of 101.3 kPa, they would amount to 164.20 to 307.88 mg/m^3 for wood biomass and up to 1847.28 mg/m^3 for corn grain.

The emissions of sulfur oxides (SO_x) and hydrogen chloride (HCl) can be significant, especially when pyrolyzing tires [241]. In studies [242], SO_2 emissions were low, ranging from 8 to 70 ppmv, even in the case of pyrolysis of materials with high sulfur content. The results of studies on gas pollutant emissions during the gasification of solid recovered fuel (SRF) in an 8-ton/day capacity installation showed NH_3 concentration in the range of 329 to 546 ppm, HCN in the range of 49 to 74 ppm, and HCl in the range of 4 to 21 ppm [243].

Typically, under technical conditions, VOC emissions can range from a few parts per million (ppm) to several hundred ppm. Carbon monoxide is highly sensitive to pyrolysis conditions and can show high variability in concentration depending on factors like temperature, pressure, residence time, and feedstock composition [244]. Studies [245] on gas emissions during the pyrolysis of wood waste showed the presence of volatile organic compounds (NMVOCs), carbon monoxide (CO), and aerosols, with modern pyrolysis units with recirculation of condensable gases reducing these pollutant emissions. Carbon dioxide (CO_2) emissions depend on the feedstock used and pyrolysis conditions. For instance, higher PET content in mixed plastic waste can lead to higher CO_2 emissions. PAH emissions can range from trace amounts to a few milligrams per cubic meter of exhaust gas [246].

5.2. Solid Waste and Byproduct Management

Effective management of ashes from municipal waste incineration processes, both bottom ashes and fly ashes, remains challenging, partly due to the presence of heavy metals [247,248]. An obvious research direction is the utilization of ashes for cement production. Cement produced from ashes shows high compressive strength and CO_2 sequestration capability [249]. Among the innovative methods of using ash in the cement industry are plasma vitrification and the inertization of ashes with silica. Plasma vitrification, especially when combined with biomass ashes, has proven to be an effective method of reducing energy costs and improving the vitrification process of MSWI fly ash, transforming it into high-quality building materials [250]. A new method utilizing colloidal silica for the inertization of MSWI fly ash has shown promising results, producing inert materials suitable for reuse in the cement industry and other applications [251].

Recent findings on the long-term leaching behaviors of heavy metals from cement containing different types of ashes confirm that heavy metals can be gradually leached over time; however, most remain within regulatory limits [252,253]. In studies [254], MSWI fly ash subjected to thermal treatment exhibited reduced leachability and transferability of heavy metals, with lower phytotoxicity and cytotoxicity compared to untreated samples, indicating safer possibilities for reuse in building materials. Current research focuses on stabilizing heavy metals in the ashes using chelating agents [255] and other techniques—for instance, studies conducted by [256] have shown that the addition of waste glass to fly ash can effectively reduce the leachability of heavy metals and improve their stabilization. In studies [257], the leaching of heavy metals was effectively reduced by sintering fly ash using microwave energy.

On the other hand, studies show that ashes from municipal waste incineration contain significant amounts of critical metals that can be recovered and used in various industrial applications [258,259].

An interesting research direction is also the use of MSWI fly ashes as materials for thermochemical energy storage (TCES). Comparative studies of fly ash from different types of incinerators (grate furnaces, rotary kilns, fluidized-bed reactors) have shown that MSWI fly ash, mainly consisting of calcium oxide (CaO), has a high energy density and meets the TCES requirements for charging, discharging, and cycle stability [260].

6. Conclusions

6.1. Summary of Key Findings

The review highlights significant advancements in the conversion of municipal solid waste (MSW) into renewable energy. The surge in urban populations has resulted in a substantial increase in MSW, demanding innovative waste management solutions. Traditional methods such as landfilling and basic incineration are no longer sustainable, given their environmental impact and inefficiency. This review emphasizes the potential of MSW as a renewable energy source through advanced thermochemical conversion technologies including incineration, pyrolysis, and gasification.

Incineration, though well-established, faces challenges related to emissions, particularly dioxins and furans. Recent innovations in high-temperature incineration and rapid flue gas cooling have shown promise in mitigating these emissions. Pyrolysis and gasification are emerging as viable alternatives, offering the ability to convert waste into energy-rich gases and oils. The integration of catalytic substances and microwave radiation in pyrolysis, along with advancements in plasma gasification, significantly enhance the efficiency and environmental performance of these processes.

Environmental impacts remain a critical concern. The review underscores the importance of stringent regulatory frameworks and advanced emission control technologies to mitigate the negative effects of waste-to-energy processes. The evolving regulatory landscape, particularly in Europe and the United States, has driven the development of cleaner and more efficient waste conversion technologies.

6.2. Recommendations for Policymakers and Practitioners

To maximize the benefits of MSW as a renewable energy source, policymakers and practitioners should consider the following recommendations:

1. Promote Advanced Conversion Technologies: Encourage the adoption of advanced thermochemical conversion technologies such as pyrolysis and gasification, which offer higher efficiency and lower environmental impact compared to traditional incineration.
2. Strengthen Regulatory Frameworks: Implement and enforce stringent emission standards to ensure that waste-to-energy plants operate within safe environmental limits. Continuous monitoring and improvement of emission control technologies are essential.
3. Support Research and Development: Invest in research and development to further improve the efficiency of waste conversion technologies and explore new methods for reducing emissions and recovering valuable materials from waste.
4. Enhance Public Awareness and Participation: Increase public awareness about the benefits and challenges of waste-to-energy technologies. Encourage community participation in waste management practices to ensure successful implementation and acceptance of these technologies.
5. Facilitate International Collaboration: Promote international cooperation and knowledge exchange to leverage best practices and technological advancements in waste management and renewable energy.

6.3. Future Research Needs

Despite the progress made, several areas require further research to fully realize the potential of MSW as a renewable energy source:

1. Optimization of Conversion Processes: Further research is needed to optimize the operational parameters of pyrolysis and gasification processes to maximize energy yield and minimize emissions.
2. Development of New Catalysts: Investigate the use of novel and low-cost catalysts to enhance the efficiency of pyrolysis and gasification while reducing the overall process cost.
3. Waste Composition Analysis: Continuously update and refine the understanding of global MSW composition to tailor waste management strategies to specific regional and economic contexts.
4. Integration with Other Renewable Technologies: Explore the integration of waste-to-energy technologies with other renewable energy sources, such as solar and wind, to create hybrid systems that can provide more stable and efficient energy solutions.

In conclusion, the transformation of municipal solid waste into renewable energy presents a promising solution to the growing challenges of waste management and energy demand. By adopting advanced technologies, strengthening regulatory frameworks, and fostering research and international collaboration, it is possible to create a sustainable and circular economy that benefits both the environment and society.

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