

## Review

# Co-Treatment of Food Waste and Municipal Sewage Sludge: Technical and Environmental Review of Biological and Thermal Technologies

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**Abstract:** To face the ongoing issues related to global warming, a circular economy approach should be pursued, rethinking the waste management system and the recovery of organic waste. The main organic waste streams are Food Waste (FW) and municipal Sewage Sludge (SS). In the spirit of circularity, a commingled treatment of FW and SS could be a viable solution. To this end, the present work aims to review the technical and environmental aspects of the co-treatment of FW and SS through biological and thermal processes. Firstly, a detailed characterization of the two substrates is presented as well as the current and future treatment technologies. Then, the technical feasibility and the environmental impacts of conventional biological co-treatments of FW and SS (i.e., composting, anaerobic digestion, and a combination of them), as well as innovative thermal ones (i.e., incineration, gasification, pyrolysis, and hydrothermal carbonization), is summarized. The outcomes of this work could contribute to achieving a more sustainable way to approach organic waste treatment and to help policy-making authorities move toward sustainable planning.

**Keywords:** food waste; sewage sludge; anaerobic digestion; biogas; composting; HTC; pyrolysis; incineration



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## 1. Introduction

An important current issue is the high energy demand from fossil fuels caused by the expanding economy and industrialization. Global warming is strongly affecting the environment, with increased average temperatures of air and sea, extreme weather phenomena, and desertification of fertile areas [1]. To face these concerns, the circular economy concept has been identified as one possible solution. In this context, the current approach to environmental management as well as the planet's resource use must be reconsidered. Waste and end-of-life products should be seen as valuable resources to close the industrial system loop [2]. For these reasons, researchers are shifting the waste management system to innovative approaches to recycling (materials, chemicals, etc.) and energy recovery.

One of the most important sources of waste is organic materials. The bulk of organic waste of municipal origin is essentially food waste or food-related waste; thus, the term food waste can be broadly used to refer to municipal solid waste of organic nature. About 1.3 billion tonnes of FW are annually produced in the world, but population growth and diet shifts are projected to stimulate a substantial increase in global food demand of 60% to 110% between 2005 and 2050 [3]. The most common FW treatment technologies are based on biological processes such as anaerobic digestion and aerobic composting. Thermal technologies such as incineration, gasification, and pyrolysis could be found, in

addition to the common disposal in landfills (which is the worst environmental practice for biodegradable waste). On the other hand, without an adequate waste management system, the pollution issue will be exacerbated in the near future [4]. The concept of managing FW with other solid waste is raising attention in countries that do not base their management on landfill.

In this context, some advantages could be associated with Sewage Sludge (SS), another waste of organic origin. Advances in wastewater treatment have improved the management of SS, producing a significant amount of this waste stream [5]. Usually, there is similarity between the treatment technologies for FW and SS, giving rise to the concept of an urban biorefinery for integrated treatment of different municipal wastes [6]. Anaerobic and aerobic stabilization are the most common procedures, but incineration and, unfortunately, landfilling, are also still carried out. However, the integration of these two waste streams would result in potential impacts and would need specific improvements to the current waste management system.

Thermal technologies such as incineration, pyrolysis, and gasification for both FW and SS are energy-intensive and do not return any organic carbon to the soil. In addition, landfill is no longer a sustainable way to treat organic waste and in general at European level it is not allowed for unpretreated waste, which makes this option the least favorable in the waste management hierarchy [7]. On the other hand, biological conversion technologies such as composting and anaerobic digestion are highly feasible technologies for general organic waste, including FW and SS.

Several reviews that deal with the co-treatment of FW and SS topic could be found in the literature. Mehariya et al. [8] summarized the recent updates and approaches to efficient anaerobic co-digestion (AcoD) of FW and SS, focusing on the scenario of Hong Kong. The impacts of FW diversion to wastewater treatment have been discussed by Zan et al. [3], identifying a FW management hierarchy. Battista et al. [9] presented the concept of FW and SS as feedstock for an urban biorefinery producing biofuels and added-value bioproducts. The concept of a biorefinery for the treatment of FW and SS has been investigated in a pilot system to produce biopolymers and biogas through anaerobic co-digestion and aerobic biomass production [10]. However, all these studies have considered either few treatment technologies or a single waste stream, without showing a comprehensive status and the technical results from the whole biological and thermal treatments of FW and SS.

In this line, the aim of this paper is to offer a comprehensive review on the co-treatment of FW and SS, focusing on the biological and thermal technologies such as anaerobic digestion and/or composting, incineration, gasification, pyrolysis, and hydrothermal carbonization (HTC). A detailed characterization of both FW and SS is provided and the specific technologies are presented. Finally, technical parameters for optimal co-treatment and the resulting environmental improvements are highlighted for each treatment.

## **2. Food Waste (FW)**

### *2.1. FW Characteristics*

Food waste (FW) is the materials intended for human consumption that are subsequently discharged, lost, degraded, or contaminated [11]. FW comprises raw or cooked food materials and includes food loss before, during, or after meal preparation in the household, as well as food discarded in the process of manufacturing, distribution, retail, and food service activities [12].

In the EU, around 88 million tonnes of FW are generated annually, with associated costs estimated at EUR 143 billion [13]. Among European countries, the United Kingdom (UK) is the one with the highest FW generation rate (more than 14 million tons), while Germany and Italy produce about 10 million tons [14]. Other countries, such as the United States, China, and Australia generate nearly 61 million, 92.4 million, and 4 million tons of FW every year [11].

Among biodegradable waste, FW is the highest one (almost 70%), followed by yard waste (less than 30%) and market waste (less than 1%) [15]. The largest amount of FW is generated during the consumption stage (46%), followed by primary production (25%) and processing and manufacturing (24%). The distribution and retail stages only account for 5% of the FW generated in the supply chain [16]. Of the FW from the consumption stage, 56.77% is avoidable, while 15.96% is from food still packed in its original packaging [17].

FW composition is extremely variable, according to the socio-economical context, as well as technical factors like waste collection systems. FW tends to vary geographically according to continent and collection source. Seasonal variation in FW characteristics can be observed in summer and winter, and during long holidays, working periods, festive seasons, etc. However, similarities in FW fractions can be pointed out, as shown in Table 1. Fruit and vegetables are the FW groups that produce the largest amount of FW (40–60%), followed by bakery (about 15%). Meat, fish, and dry food could be a significant part of FW, with great variability according to the use of the customers. Finally, dairy represents the smallest part of FW.

**Table 1.** Composition of FW according to the following sub-fractions: fruit and vegetables; bakery; meat and fish; dairy; dry food; ready meals; others.

FW SUB-Fractions	[16]	[18]	[19]	[20]	[21]	[22]
Fruit & vegetables	57%	47%	43%	41%	67%	63%
Bakery	12%	13%	18%	18%	14%	14%
Meat & fish	14%	6%	5%	7%	3%	9%
Dairy	5%	5%	7%	12%	8%	5%
Dry food	10%	6%	7%	8%	2%	
Ready meals			7%	14%		
Others	2%	23%	13%		6%	9%

Due to its variable composition, the physio-chemical characteristics of FW present a significant variability as well. However, Table 2 shows the average values of different FW key parameters.

**Table 2.** Characterization of FW according to the following parameters: humidity; volatile solids (VS); pH; total nitrogen (N) and carbon (C) content; carbon/nitrogen ratio (C/N).

Parameters	Average Values	[23]	[24]	[25]	[26]	[27]
Humidity [%]	70	72.1	69.1	84.7	77.2	72.4
VS [%TS]	80–90	94.3	85.4	85.2	88.2	68.9
pH [-]	5–6	5.0	5.2	6.0	5.1	5.4
N tot. [%]	2–4		2.2		2.8	1.8
C tot. [%]	45–55		46.5		45.5	57.2
C/N [-]	15–30	18.1	21.1	15.7	18.5	31.8

The solid content of FW is about 50%, which is higher compared to other organic substrates such as sludge. This reflects on the limitation of the loading capacity during processing. The solid fraction is composed mainly of degradable solids, especially lipids in the case of the presence of animal fat and oil. On the other hand, FW is characterized by low C/N and pH, which range between 15–30 and 5–6, respectively. Indeed, fruit and vegetables are characterized by high levels of nitrogen, as opposed to carbohydrates such as potatoes and bread, which have a significant amount of carbon [25].

## 2.2. FW Treatment Technologies

Several industrial-scale technologies are available for FW treatment, ranging from waste-to-energy (WtE) for energy production to material recovery [11]. The most common treatments of FW are anaerobic digestion and composting, followed by incineration. Landfilling represents the least desirable option. However, other technologies are emerging.

### 2.2.1. FW Biological Treatment Technologies

**Composting:** Composting is a consolidated technology for FW treatment. It consists in the aerobic degradation of the putrescible fraction to reduce the waste volume and to recover nutrients through the formation of humic substances. Composting is then an economically feasible and technically reliable technology for the recovery of material (the compost) in further use in agriculture or floriculture [28]. At the industrial level, the process takes place in a closed environment with controlled temperature, humidity, and oxygen content. Industrial composting consists in a bio-oxidation step of few weeks (2–3) characterized by thermophilic temperatures (which ensure the sanitation of the biomass), followed by a curing step of 2–3 months at lower temperatures [29]. The feasibility of FW composting is widely known but, as with other substrates, it requires the adoption of bulky materials such as garden waste, wood chips, wheat straw, or sawdust. Bulking agents give structure and porosity to the mixture for proper aeration, but also absorb part of the leachate produced during the decomposition process [30]. The adoption of bulking agents is fundamental especially for the treatment of FW, due to its fast degradation. On the other hand, composting has the limitations of (i) requiring long treatment times; (ii) producing odours and greenhouse gases like CO<sub>2</sub>, N<sub>2</sub>O, and NH<sub>3</sub> (which are attributed to degradation of proteins and lipids present in FW); (iii) generating leachate and other waste; and (iv) requiring significant amounts of energy [31].

**Anaerobic digestion:** Anaerobic digestion (AD) is another very common technology for the treatment of FW, which started to gain attention at the end of 1990s. AD is a WtE process, which aims to produce a biogas used in thermal and electric energy production. The biogas from FW is rich in methane (60–70%), with the remaining part composed mainly of CO<sub>2</sub> and other gases (N<sub>2</sub>, H<sub>2</sub>, H<sub>2</sub>S and NH<sub>3</sub>) in trace amounts [32]. In addition to the biogas, AD can produce a digestate rich in nutrients for land application. Compared to composting, AD requires less space and generates less odour, while the greenhouse gases emitted are collected without any spread in the atmosphere [33]. Despite AD requiring energy as well as composting, the energy recoverable from the process could result in net energy production. Due to its composition, FW causes high methane production (300–600 NmlCH<sub>4</sub>/gVS [34]) as well as high biodegradability and vs. reduction [35]. On the other hand, the loading rate of FW in AD should be low, since high loading is associated with digestate acidification from the accumulation of inhibitory compounds such as VFA, mainly caused from FW's extremely high degradability [36,37]. Inhibition effects were observed at a high loading rate of 2.5 gVS/L [38] or when the total ammonium concentration in the reactor exceeded 2 g/L [39]. Other causes of process failure are inhibition of methanogenic pathways due to an improper C/N ratio and trace elements limitations [40]. In addition to the process stability issues, AD presents some limitations like the small reduction in volume of the inlet waste, the necessity to collect the biogas to not affect the environment (CH<sub>4</sub> has an impact 20 times higher than CO<sub>2</sub> [41]), and the use of the digestate (which is often too acidic for any land application). AD of FW could be also used to produce specific chemical compounds such as precursors for plastic material production, and chemical or pharmaceutical applications.

**Dark fermentation:** From AD, it is possible to generate not only methane but also hydrogen (which has double the energy yield of methane) through the dark fermentation process. The high protein levels in FW reduce the hydrolysis rate as well as the hydrogen yield [42]. For this reason, dark fermentation is usually implemented in a two-stage anaerobic digestion system. The first stage is usually maintained at an acidic pH of around 5.5–6.5 and a short hydraulic retention time (HRT; 2–3 days) for acid fermentation, and the

second stage is usually operated at an HRT of 20–30 days and pH of 6–8 to facilitate the proliferation of slow-growing methanogenic archaea [43].

*Combined anaerobic digestion–composting:* Composting can be used to treat residues from industrial anaerobic digestion processes (e.g., digestate). Combining anaerobic digestion and composting can constitute a feasible method not only to recover energy, but also to improve the quality of the compost, reducing the issues related to digestate such as odour emission, the concentration of volatile compounds, the moisture content, the potential phytotoxicity, and the presence of pathogens [44]. Thus, a post-treatment to anaerobic digestate is advisable to enhance digestate properties prior to its agricultural use as well as the composting process itself. A post-treatment can consist of a solid–liquid separation of the digestate. The liquid fraction can be recirculated in the anaerobic digestion process, managed using depuration treatments, or used directly for agricultural purposes [45]. About 50% of the initial nitrogen and 86.4% of the initial phosphorus can be found in the final compost, which achieves a high level of stabilization and a reduction of 93% from that of the raw inlet waste (mainly occurring during anaerobic digestion) [46]. The combined anaerobic digestion and composting process utilizes the advantages of the two processes. Despite the spread of this configuration, usually costs remain high.

## 2.2.2. FW Thermal Treatment Technologies

*Incineration:* Incineration is another example of a WtE process for FW, which, however, is rarely adopted at the industrial scale. Indeed, due to its humidity, FW has a low higher heating value (27.92 MJ/kg dry FW [47]) and is usually co-incinerated with other waste. However, with incineration, a large volume of FW could be reduced (80–85% of the incoming waste [48]), but European countries are still reluctant to adopt this technology due to its release of toxic air emissions (e.g., dioxins, heavy metals, etc.) [49]. Other possible thermal processes for FW treatment are pyrolysis and gasification: the main differences between these technologies are the combustion environment (oxygen-free and partially oxygenated, respectively) and the temperature adopted (300–700 °C and 800–900 °C, respectively). Their outputs are oils, char, and gas for energy recovery. Not only due to the high humidity of FW but also for the cost of the treatments, pyrolysis and gasification are not currently adopted as FW treatment [50]. However, they remain a suitable technology for FW since a syngas composed mainly of CO and H<sub>2</sub> (85%), with a small proportion of CO<sub>2</sub> and CH<sub>4</sub>, could be obtained [51].

*Hydrothermal carbonization:* To face the issue related to the substrate humidity in thermal processes, hydrothermal carbonization (HTC) has been developed. HTC is a wet process that converts FW to a valuable, energy-rich resource (15–30 MJ/kg dryFW) under autogenous pressures and relatively low temperature (180–350 °C) [51].

## 2.2.3. FW Disposal

Finally, landfilling is the last but, unfortunately, still the most adopted FW management option, especially in developing countries. Due to the semi-anaerobic environment, FW degrades in landfills, producing CO<sub>2</sub> and CH<sub>4</sub>, which, if not collected, have a strong impact on the atmosphere [52]. In addition, landfilling generates issues like the rising cost of waste disposal, the lack of land space, and groundwater pollution by leachate [36]. For these reasons, the “Landfill Directive” n. 850 of 2018 from the European Parliament has forbidden the disposal of biodegradable waste in Europe without any pre-treatment.

## 3. Sewage Sludge (SS)

### 3.1. SS Characterization

Sewage sludge (SS) is the residue resulting from the treatment of wastewater released from various sources, including homes, industries, medical facilities, street runoff, and businesses [53]. Municipal SS is defined as the final solid residue produced during municipal wastewater treatment. It is classified as a solid waste with the code of 19 08 05 according to the European Catalogue of Wastes (EEL 47/16-2-2001; Directive 2000/532/EK) and it has to



be treated as a proper waste stream according to the Wastes Framework Directive applied since 12 December 2010 [EU, Directive 2008/98/EC]. In general, a conventional wastewater treatment plant produces two types of SS: primary sludge is the solid accumulated from the physical separation process and consists of gravitational precipitates; secondary or activated sludge is the byproduct generated from the biological treatment plant, containing a high amount of active microbes [54].

The amounts of sludge produced in the European Union (EU) are very large and have increased from 6.2 million tons in 2010 to more than 9 million tons in 2020, with Italy, the United Kingdom, Belgium, France, and Germany being the largest producers [55]. However, due to a lack of homogeneity and the fragmentation of the data, a clear quantification is still not possible, and values could range widely according to the database consulted [56].

SS mainly consists of organic matter and nutrients, but its content is highly variable, since it depends on the SS treatment process and the seasonality [57]. An SS characterization according to the main physio-chemical parameters is shown in Table 3.

**Table 3.** Characterization of SS according to the following parameters: total solids (TS); volatile solids (VS); pH; chemical oxygen demand (COD); total organic carbon (TOC); total nitrogen (N) and phosphorous (P) content; carbon/nitrogen ratio (C/N); heavy metal contents.

Parameters	Primary SS		Secondary SS	
	Raw	Digested	Raw	Digested
	[58–63]	[60,62]	[58,60,61,63]	[58,63,64]
TS [%]	2–8	6–12	0.5–3	20–30
VS [%TS]	60–80	30–60	55–75	30–50
pH [-]	5–8	6.5–7.5	6.5–8	7–8
COD [mg/L]	1750		4195	1518
TOC [%TS]	30–50		50–55	50
N tot. [%TS]	1.5–5	1.6–6	2.4–6	5
C/N [-]	10–20		10–25	10
P tot. [%TS]	0.6–3.5	1.5–4	0.5–1.5	2–4
As [mg/kg] *			2–9	
Cd [mg/kg] *			0.5–1.5	
Cr [mg/kg] *			20–80	
Cu [mg/kg] *			180–390	
Hg [mg/k] *			0.5–1.5	
Ni [mg/kg] *			17–50	
Pb [mg/kg] *			30–80	
Se [mg/kg] *			2.5–4.5	
Zn [mg/kg] *			400–1000	

\*: heavy metal values are from Mininni et al. [65].

SS is characterized by a poor dewaterability and a high moisture content (which could differ according to the source or the preliminary digestion). Based on the source (primary or secondary), the level of volatile solids could change, as well as the pH. SS results in low C/N ratios and high buffer capacity, which may also cause odour emissions when composted [66]. The contamination of SS by heavy metals is widely known; high levels of zinc and copper can be found; elements like chromium, lead, and nickel are also evident; and selenium, arsenic, cadmium and mercury can be found in trace amounts [65].

Primary SS contains more carbohydrates than secondary SS (about 45% and 20%, respectively). On the other hand, secondary SS is higher in proteins than primary SS (about 50% and 20–30%, respectively), though they have similar amount of lipids (less than 10%) [67].

### 3.2. SS Treatment Technologies

To prevent environmental issues related to air emissions, threats to public health, and contamination of soil and water resources, municipal SS therefore requires an appropriate treatment and careful management.

#### 3.2.1. SS Reuse

A high content of organic compounds with macro- and micronutrients, especially nitrogen and phosphorus, makes SS an excellent fertilizer, a cheap and rich soil enhancer. For this reason, the direct use of SS in agriculture is usually adopted in several European countries [68]. However, the microbiological activity of SS affects its stability, resulting in a low germination rate and thus restricting its use in agriculture [53]. In addition, the high level of heavy metals as well as other undesirable and dangerous substances such as hormones, antibiotics, and pharmaceuticals pose a huge risk for the ecosystem [69]. The application in agriculture of SS is regulated by the “Sludge Directive” 86/278, which is based on limits for specific heavy metals (Cd, Cu, Hg, Ni, Pb, and Zn). Some countries have imposed more stringent limits than those of the directive. On the other hand, there is still no uniform approach on the limits for organic micro-pollutants, for which only few countries have adopted some regulations [70].

#### 3.2.2. SS Pre-Treatment

When SS is produced, a stabilization process is performed, usually directly in the wastewater treatment plant, which acts as a pre-treatment. The typical SS stabilization is composed firstly of a sludge thickening or dewatering through different techniques (centrifugation; gravity settling; rotatory drum filtration; air flotation) to increase the TS concentration. Then, a degradation of the SS volatile solids and a reduction of pathogenic organisms and other unwanted qualities of the sludge is carried out in an aerobic environment through biological stabilization [58]. Stabilization could be also carried out under an anaerobic environment, through AD. Factors influencing SS stabilization are the presence of heavy metals (which can cause toxicity to microorganisms) and a limited availability of microorganisms able to degrade a large variety of contaminants [58]. In addition, these treatments are characterized by a limited efficiency, usually difficult maintenance, and high costs, which accounts for almost 50% of the total wastewater treatment plant costs [71]. For these reasons, further processes are necessary to reduce the impact of SS on the waste management system.

#### 3.2.3. SS Biological Treatment Technologies

*Composting:* As with FW, composting is a suitable option for SS treatment, currently adopted at full-scale. The high temperature reached due to the metabolic heat generated during the thermophilic phase of the composting process could effectively destroy pathogens and enhance biological degradation of different organic micro-pollutants [72]. Especially for the treatment of SS (which has high moisture content and low porosity), it is necessary to use bulking agents to create an aerated matrix [73]. Composting reduces the fecal coliform level of SS, decreases its C/N ratio, and immobilizes ammonia in organic forms [64]. However, SS has too little organic content to achieve thermophilic conditions during composting, which is necessary for complete pathogen destruction [74]. In addition, SS composting is usually associated with VOC, NH<sub>3</sub>, and H<sub>2</sub>S emissions, which are generally associated with odorous nuisance and health risks [75]. The odor concentration of SS could be reduced by 40% during composting by increasing the time of composting, until 89% if the curing piles are turned [76].

*Anaerobic digestion:* AD can reduce the vs. of SS by 35–60%, while TS reduction does not exceed 30% [59]. The high level of carbohydrates, lipids, and proteins in SS allows for a potential methane production of about 300 L/kg VS, ranging from a minimum of 175 L/kg vs. to a maximum of 475 L/kg vs. [67]. AD is carried out mainly in mesophilic conditions (30–42 °C), which ensures stabilization but not sufficient hygienization. Thermophilic fermentation (50–60 °C) gives better results in lowering the level of pathogens, as well as shortening the digestion time and increasing the methane production. However, the thermophilic digestion process, compared to mesophilic digestion, is more sensitive to small changes in the process parameters and requires more energy to heat the SS to the required temperature [62]. Pre-digested SS has a residual amount of degradable materials lower than raw SS, resulting in a limited biogas production. For this reason, AD is mainly adopted on raw SS or on a mixture of digested SS with other organic substrates through a co-digestion process [77]. High-solid anaerobic digestion of SS is a promising alternative to conventional AD as it reduces treatment volumes, transportation costs, and energy consumption for heating, and increases the fertilizing potential of SS. Nevertheless, several challenges are posed for the structural and rheological characteristics of dewatered SS and to toxic compounds originating during and before the anaerobic treatment [78]. In general, AD of SS tends to be slow and unstable due to nutrition deficiency, low organic loading rate, lows biodegradability, and the high toxicity of contaminants [79].

*Combined anaerobic digestion–composting:* The viability of combined anaerobic digestion and composting of SS was also reported. Indeed, a substantial decrease in the C/N ratio, pH, VS, total organic carbon, and pathogenic populations were detected in the compost obtained after anaerobic digestion, while a significant increase was observed in the ash and nitrogen contents. These achievements confirmed an adequate degree of maturity and stability of the compost obtained, reflecting its suitability for agricultural use [80]. Using a combined treatment is advantageous for both municipal and industrial SS due to minimal space requirements, low capital cost, and excellent COD removal efficiencies [81].

### 3.2.4. SS Thermal Treatment Technologies

*Incineration:* Incineration is becoming the most popular SS utilization because of the significant reduction of volume (about 10%) for the thermal decomposition of organic matter [62]. Due to an elevated moisture level, SS is combusted with other waste with higher heating value (e.g., municipal solid waste). Despite this, incineration and co-combustion are the most common disposal routes in some EU-15 countries; these technologies encounter high investment costs and are not always welcomed by citizens [56]. SS incineration for electricity production could achieve environmental benefits for human toxicity, freshwater aquatic ecotoxicity, acidification, and eutrophication, while it has great negative effects on environmental categories such as global warming and ozone layer depletion [82].

*Pyrolysis:* Compared to other thermal treatments, pyrolysis seems to be the optimal thermochemical treatment option since it is favourable for energy savings, material recovery, and high added materials production, providing a ‘zero waste’ solution [63]. In addition to the conversion of organic matter to biofuels and high-quality biochar, pyrolysis can significantly minimize the SS volume, kill pathogens, and immobilize heavy metals in the biochar matrix [83,84]. SS could be rapidly processed from 500 °C to 900 °C [85], but the biochar yield is maximized at 300 °C [86].

*Gasification:* The gasification process can be used for SS with the TS in the range of 70%–95%. This process has a huge potential for reuse of waste as an energy source and it can be valuable mainly for the cheap raw materials such as SS. The energy accumulated in the produced gas is highly dependent on agents such as either the feed fuel or reaction type, but also many others. In comparison to the incineration, gasification does not need any other fuels as supplements, and the emission of toxic gases such as sulfur dioxides and nitrogen oxides is reduced. The highest disadvantage of SS gasification is the problem with the concentration of heavy metals in the final product [62].



*Wet oxidation:* An example of a chemical oxidation process alternative to incineration, wet oxidation allows for a drastic reduction in the organic matter in SS, converting it into a gaseous phase (consisting of CO<sub>2</sub> and H<sub>2</sub>O) and a liquid one (sent to a wastewater treatment plant for further processing) [87]. This technique can be applied to SS with a TS between 1% and 6% (a preliminary dewatering process is then necessary) at certain temperature conditions (150–360 °C), oxygen concentration (or air), and pressure (30–250 bar), for a continuous contact time of 15–120 min [71]. Some applications at a real scale are currently available thanks to numerous patents, which allow the removal of 80–97% of VS and 43–71% of COD, depending on the operating conditions [61]. On the other hand, wet oxidation has the advantage of requiring relatively low investment and low energy demand, but it requires highly qualified personnel [88].

*Hydrothermal carbonization:* HTC could be adopted for the treatment of SS since it can simultaneously increase the dewaterability of SS, biologically sterilize it, and separate the liquid by-product from the chars [89]. The heating value of SS could be enhanced 1.02–1.10 times more than that of raw SS if HTC is carried out at 260 °C for 30–90 min [90].

### 3.2.5. SS Disposal

Finally, landfilling is unfortunately the most common disposal method in several countries [68]. In a landfill, SS degrades three times slower than in composting [59], and it affects the leachate production as well as the CO<sub>2</sub> emission. Landfilling of SS (or generally biodegradable waste) should be avoided in favor of other treatment options, as it is not an efficient environmental approach. Indeed, the amount of SS disposed of in landfills is intended to rapidly decrease in upcoming years, since the Landfill Directive (EC Directive, 1999) obliges the member states to reduce the amounts of biodegradable waste (and therefore of SS, which is a biodegradable waste) sent to landfills to 35% of 1995 levels by 2016.

## 4. Biological Co-Treatment of FW and SS

As pointed out in the previous section, advances in FW and SS treatment have been developed to meet the need for a more sustainable society by improving their resource recovery of water, chemicals, and energy. Thanks to the existing technologies which FW and SS have in common, diverting SS from the wastewater stream to the solid waste one is promising for improving treatment efficiency, deriving economic benefits, and abating environmental impacts [3]. However, integration of FW with SS could potentially affect the management systems. In this context, this section offers a comprehensive analysis of the effect of a combined FW and SS treatment, focusing on the biological processes of composting and/or anaerobic digestion. Tables 4–6 summarize several works dealing with the topic of FW-SS co-composting, co-digestion and combined co-composting and co-digestion, respectively, highlighting the substrates tested, the process parameters, and the main results achieved.

### 4.1. Composting

Composting relies on a decomposition of the organic matter by a diverse community of microorganisms, including bacteria, fungi, and actinomycetes, according to the stages of the process. Composting starts with an initial mesophilic phase of few days where mesophilic microorganisms break down readily degradable organic materials. The degradation increases the temperature, leading to a thermophilic phase where complex organic compounds like proteins, fats, and cellulose and pathogens are killed by the elevated temperature. As the easily degradable compounds are exhausted, the temperature gradually decreases, restoring a second mesophilic phase. This phase is followed by a maturation of some months, in which humic substances are formed, aiming to produce the compost. The composting process typically starts with a pH of around 5.0–6.5 and gradually increases to neutral (7.0) as the compost matures.

Co-composting of SS and FW could produce benefits from both the technical and environmental points of view. Co-composting could minimize the SS composting disadvantages (high heavy metal contents, low C/N ratio, and high moisture level) by diluting heavy metal concentrations, increasing the carbon and nitrogen contents, and allowing the moisture of the mixture to be adjusted to optimal values [91]. On the other hand, the use of FW (characterized by acid pH) in combination with SS can balance the pH of the mixture [92]. The optimal presence of FW in the mixture ranges from 55% to 66% by weight: in this proportion, the final compost is improved in N, while the process achieves earlier thermophilic temperatures for a longer time [93,94]. In addition, many bacteria capable of degrading organic matter were detected in co-composting when the FW content was higher [95]. The use of FW such as olive mill waste was beneficial since it increased the phenol level, causing a significant decrease in pathogens [96]. *E. coli* was not detected in compost from the combined process, making it suitable for the culture of edible plants [74,96]. This suitability could be affected by an elevated (55–85%) presence of FW in the mixture as well as the compost quality in terms of salinity [94]. Mixing SS and FW up to a 25 C/N ensures high nutrient levels, microbial diversity and richness, and low biosecurity risk [92].

In addition, co-composting of FW and SS can effectively remove antibiotics, fluoroquinolones, and macrolides (mainly present in SS), while showing poor efficiency in removing sulphonamides [69]. Indeed, SS contains various classes of organic pollutants such as polyaromatic hydrocarbons (PAHs), polychlorinated biphenyl (PCBs), polychlorodibenzo-p-dioxins and polychlorodibenzofurans (PCDD/Fs), phthalates (PAEs), perfluorinated compounds (PFCs), and pharmaceuticals and personal care products (especially antibiotics) [97]. Co-composting can effectively remove some organic pollutants such as PAEs and PAHs. Nevertheless, some halogenated organic contaminants (e.g., PCDD/Fs, PCBs, and PFCs) are recalcitrant to be degraded during sludge composting, resulting in low removal or even elevated levels due to organic matter decomposition. Moreover, the disadvantage of long composting duration (e.g., >100 d) with low dissipation of organic pollutants limits the amount of sewage sludge that can be composted [98].

The composting of organic waste, especially at high moisture levels of the pile (in the case of SS–FW co-composting), produces significant amounts of leachate. This liquid generated from the decomposition of organic compounds depends on various factors such as the composition of the waste, but in general has a tendency of adsorbing a wide variety of contaminants (which implies a proper management of such output) [99].

Co-composting of FW and SS is more favourable in terms of odour emission since it could reduce almost 55% of the emission rate during the process [91]. Even for a small concentration of eggplant waste (4.7% and 8.6%), the odour emissions could be reduced up to 34.6% and 63.6%, respectively, compared to the composting of only SS [100]. The CO<sub>2</sub> emission from co-composting is comparable to that obtained from the composting of FW only [94]. In addition, N<sub>2</sub>O emission from co-composting are 18–26% lower than that obtained by mono-composting: N<sub>2</sub>O is not only a greenhouse gas with a global warming potential that is 310 times greater than that of CO<sub>2</sub>, but also contributes to ozone layer depletion [101].

For all these reasons, co-composting of SS and FW finds industrial application in several countries. In Japan, 20 of 38 composting facilities were characterized by a co-treatment of these streams [102], while in Italy, 18 plants are currently co-composting SS and FW [15].

**Table 4.** Literature review of composting processes of FW and SS in terms of substrates tested, process parameters, and results achieved.

Reference	Substrates Tested	Process Parameters	Results
[93]	Organic waste from screened MSW (60 mm dimension; 38.8% TS; 29.6% C <sub>org</sub> ); Dewatered SS (22.2% TS; 26.5% C <sub>org</sub> ); Mature compost	Composting process of 28 days varying the proportion of the substrates tested, the aeration pattern (continuous and intermittent), and rate (0.2–0.8 L/min kgVS).	Continuous aeration of 0.5 L/min kgVS ensured a faster initiation and maintained moderate moisture content for microorganisms, compared to an intermittent aeration pattern. A 3:1 (v:v) mixture of organic waste and SS was most beneficial to composting since it maintained higher temperatures for longer duration, achieved the fastest organic degradation, and resulted in a compost with higher N content. Further benefits could be obtained by the addition of fresh compost in the mixture by a 3:1:1 (v:v:v) proportion of organic waste, SS, and compost.
[101]	Organic waste from MSW sorting (20 mm; 39.8% TS; 32.8% C <sub>org</sub> ; 29.1 C/N); Dewatered SS (61.9% TS; 25.3% C <sub>org</sub> ; 22.6 C/N)	Three composting windrows on full-scale with forced aeration for 32 days at different SS concentrations (0, 25, and 50%).	The use of SS during organic waste composting reduced the total N <sub>2</sub> O amount by 18.4–25.7% compared to that from mono-composting (only organic waste). The emission was higher during the initial stage, decreasing during the cooling and maturation stage due to, the abundance of denitrifying bacteria.
[91]	Organic mixture composed of SS (61.9% TS; 25.3% C <sub>org</sub> ; 22.6 C/N), strawberry extrudate (61.9% TS; 25.3% C <sub>org</sub> ; 22.6 C/N) and fish waste (61.9% TS; 25.3% C <sub>org</sub> ; 22.6 C/N) in ratio of 190:1:22 (dry weight)	Six days of active composting in an adiabatic reactor (124-L capacity) filled with 14 kg of waste, airflow rate of 185 L/h, and oxygen concentration of 16–20%.	Co-composting of organic waste mix reduced the organic matter concentration (in terms of vs. and %C) by about 15%. The odor emission during co-composting was 55% lower than that generated during composting of only organic waste.
[74]	Organic mixture composed of SS (34.3% C <sub>org</sub> ; 11.1 C/N), oil palm empty fruit bunches (64.6% C <sub>org</sub> ; 43.5 C/N), and cocoa pod husks (41.1% C <sub>org</sub> ; 25.1 C/N)	Three weeks of composting in piles at full-scale at different mixture ratios, with moisture monitoring and pile turning at regular basis.	Maximum temperatures ranged 46.8–54.5 °C during co-composting, avoiding the presence of <i>E. Coli</i> at the end of the process. The mixture of SS, oil palm empty fruit bunches, and cocoa pod husks in a ratio of 2:2:1 was found to be the safest formulation, suitable for the growth of tomato plants.
[94]	Organic fraction of MSW (33.5% TS; 35.1% C <sub>org</sub> ; 16.0 C/N) consisting of different FW fractions (57.9% vegetables, 12.7% peels, 13.7% staple food, 4.2% meat, 6.3% eggshells, bones and shells, and 5.3% nutshells and cores; wet basis); SS (16.9% TS; 25.2% C <sub>org</sub> ; 6.4 C/N) Cornstalk (91.5% TS; 43.9% C <sub>org</sub> ; 52.9 C/N)	Seven 15-day composting treatments in a 60 L stainless steel cylinder with monitoring of temperature and forced aeration (0.4 L/min kgDM), manually turned every 3 days. FW proportions from 0% to 85% were investigated.	FW should not exceed 55% in co-composting with SS in order to balance the rapid initialization of the process (due to the high SS proportion), the longer thermophilic phase, and the higher humification degree (due to the high FW proportion). Excessive FW required a longer co-composting period to ensure a desirable compost maturity and quality in terms of salinity and plant toxicity.
[100]	Eggplant waste (10 mm dimension; 89.9% TS; 2.7% C <sub>org</sub> ; 42.6 C/N); Mixture of SS (20.2% TS; 5.7% C <sub>org</sub> ; 18.9 C/N); Wood chips (82.7% TS; 1.0% C <sub>org</sub> ; 96.4 C/N)	Six composting piles at full scale for 86 days turned regularly with SS-wooden chip ratio of 1:3 and small content of eggplant waste (4.7% and 8.6%).	During the hydrolytic stage of the co-composting process, the odor concentration was lower when the eggplant waste content was higher (6317 and 8192 ouE/m <sup>3</sup> ) in comparison with the lower concentration (9214 and 14,720 ouE/m <sup>3</sup> ) or without its addition (reference composting pile: 10,200 and 22,500 ouE/m <sup>3</sup> ).

Table 4. Cont.

Reference	Substrates Tested	Process Parameters	Results
[95]	FW and SS	Lab-scale composting system of three SS:FW ratios (1:1; 2:1; 4:1) at forced aeration (0.2 L/min kg).	An SS:FW mass ratio of 1:1 during co-composting enhanced nitrogen fixation ability and ammonia nitrogen level while the abundance of bacteria was increased.
[96]	Olive mill waste (29–68% TS; 66.3–68.4% C <sub>org</sub> ; 63.4–73.3 C/N); SS (42% TS; 26.4% C <sub>org</sub> ; 9.4 C/N); Green waste (40% TS; 55.0% C <sub>org</sub> ; 70.5 C/N)	Two piles of composting cycles for 60 days at different substrate ratios, regularly turned and moistened.	Co-composting allowed obtaining hygienic compost with sufficient agronomic quality for direct agriculture use such as P and K, which met similar quality compared to commercial composts. Phenol accumulation from the mixture containing olive mill waste caused an important decrease in pathogens within the compost. Compost application in peat amended at ratios equal to 30% and 50% improved the growth speed and fresh biomass of maize and tomato plants.
[92]	FW (22.0 C/N), SS (7.8 C/N) and rice husk	Four lab-scale composting reactors (composting of FW; composting of SS; SS-FW co-composting at a C/N of 25; SS-FW co-composting at equal proportions) for 47 days with 55% moisture maintained and regular turning.	Among the four piles, co-composting at 25 C/N degraded effectively polysaccharides and proteins, achieving the highest level of total nutrient as well as the highest microbial diversity and richness. The biosecurity risk of co-composting at 25 C/N was lower than that of single-composting.

Table 5. Literature review of anaerobic digestion process of FW and SS in terms of substrates tested, process parameters, and results achieved.

Reference	Substrates Tested	Process Parameters	Results
[103]	FW (21.4% TS; 92.8%TS VS; 4.7 pH); Dewatered SS (20.4% TS; 56.7%TS VS; 7.5 pH)	Five stirred reactors of 6 L volume, at 35 °C, with solid retention time of 8–30 days, at different SS:FW mixing ratios (from 2.4:1 to 0.4:1).	The addition of FW improved system stability and greatly enhanced volumetric biogas production. The addition of SS reduced Na <sup>+</sup> concentration and helped maintain satisfactory stability during the conversion of FW into biogas. Biogas production and vs. reduction in digestion of the co-mixture of SS and FW increased linearly with higher ratios of FW.
[104]	FW (14.5% TS; 95.4%TS VS; 8.2 pH) Secondary activated SS (5.0% TS; 75.7%TS VS; 6.9 pH)	Three digesters of 4.5 L each, at 30 °C, with HRT of 20 days for 90 days, and at FW:SS of 3:1, incrementing the OLR from 0.5 to 6 gVS/Ld.	At an optimum FW:SS of 3:1, high OLR (6 gVS/Ld), and intermittent biogas recirculation (2000 mL/min for 15 min/h produced the maximum biogas (0.28–0.86 L/gVSr). The synergistic effect of CO <sub>2</sub> acidification and high VFA production led to the benefit of reduced digester pH (from 8.3 to 6.6), NH <sub>3</sub> control (2380 mg/L), and in-situ CH <sub>4</sub> enrichment (88%).

Table 5. Cont.

Reference	Substrates Tested	Process Parameters	Results
[105]	FW (18.3% TS; 94.4%TS VS; 5.2 pH) Bio-flocculated SS (<1% TS; 51.9%TS VS; 7 pH)	A 6.5 L reactor at 37 °C with a HRT of 4 days and OLR of 2.5 gVS/Ld with a SS:FW of 98:2.	The highest VFA accumulation of 1902 mg/l were observed, and maximum bio-methane yield was found to be 127.05 mLCH <sub>4</sub> /g VSa. The pH with current feeding ratio was found stable (between 6.5 and 7.5) during the reactor operation without adding an external alkalinity source.
[106]	Raw FW (3.0% VS; 5.8 pH) Pre-fermented FW (3.7% VS; 4.4 pH) SS (1.8% VS)	A 400 L digester working at 30 °C, HRT 30 d, and OLR 0.4 gVS/Ld, having as feedstock SS:FW 80:20% (v/v)	Pre-fermented FW co-digestion of FW (either raw or pre-fermented for 48 h at 24 °C) and SS presented better performance (53% vs. reduction and 186–223 NmL biogas/g VSa) than the SS mono-digestion fed with only sludge (35% vs. reduction and 41 NmL biogas/g VSa). The digester fed with SS and pre-fermented FW achieved 1.5 times faster pH recovery (occurred in the first weeks of operation) and relatively stable biogas production throughout the operation.
[107]	FW (9.1% TS; 87.3%TS VS; 5.3 pH) SS (2.1% TS; 38.0%TS VS, 7.5 pH)	A 3 L reactor working at 35 °C, varying the OLR from 1.5 6 g-TS/Ld on.	High OLR of 12.6 g-TS/Ld and short HRT of 7.5 d effectively enhanced co-digestion of FW and SS, preventing the instantaneous feeding shock to the digestion system and creating an adaptable environment for the microbes, which increased methanogenic capacity and CH <sub>4</sub> yield.
[108]	FW (17.4% TS; 92.5%TS VS; 6.5 pH); Dewatered SS (1.3% TS; 84.6%TS VS; 6.0 pH)	BMP tests were performed in a 1 L glass reactor at 35 °C at 1:1 ratio (VS basis).	Co-digestion of primary SS with FW resulted in higher specific methane yields (799 mL/g VSa) than that from mono-digestion (159 and 652 mL/g VSa for SS and FW, respectively). No substantial differences could be found in process stability parameters such as pH, ammonium-N, and volatile fatty acids, which were, for all the samples, in the range of 7.9–8.1, 480–830 mg/L, and <600 mg/L, respectively. VS and COD removals were found to be 117.8% and 127.1%, respectively in co-digestion, very far from the one obtained in SS (75.8% and 83.7%, respectively) and FW (93.4% and 87.1%, respectively) mono-digestion.
[109]	Diluted FW (5.2% TS; 90.9%TS VS; 4.4 pH); SS (4.5% TS; 49.5%TS VS; 6.8 pH)	BMP tests were conducted in 120 mL glass serum bottles at 37 °C for 60 days. Seven groups of tests with different SS:FW ratios were carried out: 1.0:0.0 (mono-digestion of SS), 0.8:0.2, 0.6:0.4, 0.5:0.5, 0.4:0.6, 0.2:0.8, and 0.0:1.0 (mono-digestion of FW) at vs. basis.	The SS:FW ratios of 0.5:0.5 showed an increase in methane productivity of 4.59 times (50.30 mL/gVS d), a reduction in lag-phase shortening of 11.53 times (0.182 day), and an increase in hydrolysis rate of 3.88 times (0.334/day) compared with the SS mono-digestion.



Table 5. Cont.

Reference	Substrates Tested	Process Parameters	Results
[110]	FW (19.9% TS; 92.0%TS VS; 4.4 pH); SS (1.0% TS; 83.6%TS VS; 6.8 pH)	BMP tests were performed in a 0.5 L glass reactor at 35 °C at different FW:SS mixing ratios (1:0, 1:1, 2:1, 1:2 and 0:1; vs. basis).	The biggest synergistic effect happened in the co-digestion of SS and FW in the vs. mixing ratio of 1:1, with the highest methane yield of 415.3 mL/g VSa.
[111]	FW (23.9% TS; 91.2%TS VS); SS (16.9% TS; 57.4%TS VS); Garden waste (97.3% TS; 91.9%TS VS)	BMP tests were performed for each substrate in a 0.5 L glass reactor at 37 °C for 60 days. Semi-continuous reactors (0.5 L; 4.0 gVS/L day; 20 days) of different substrate combinations were carried out.	Co-digestion of SS and FW showed improved process stability and archaea/total microbe ratio (from 0.4% of FW mono-digestion to 17.1%), which might be due to the regulating effect of abundant trace metals in SS. The co-digestion of SS, FW, and yard waste resulted in high methane yields of 314.9 mL/g VSa with a reliable stability.
[112]	Diluted FW (4.2% TS; 97.0%TS VS); SS (3.0% TS; 49.0%TS VS)	BMP tests were performed in a 160 mL glass reactor at 35 and 55 °C at different FW content (from 0 to 80%; vs. basis) and density (from 1 to 4 g VS/L)	Thermophilic conditions (55 °C) lead to a biogas production of 215 L/kgVS compared to 157 L/kgVS under mesophilic conditions (35 °C), which corresponded to 85.3 and 35.3% biogas increases, respectively, compared to FW mono-digestion. Optimal mixing ratios of FW were 39.3% and 50.1% in mesophilic and thermophilic conditions, respectively.
[113]	FW from different sources (canteens, supermarkets, restaurant, household, fruit and vegetable markets, and bakery); SS (3.5% TS; 65.5%TS VS; 5.3 pH)	BMP tests were conducted at mesophilic temperature both on single substrates and in co-digestion regimes with different substrate mixing ratios.	The maximum methane yield was observed for restaurant (675 NmlCH <sub>4</sub> /gVSa) and canteen organic waste (571 and 645 NmlCH <sub>4</sub> /gVSa). The best co-digestion BMP test has highlighted an increase of 47% in methane production with respect to SS digestion. An equal amount of FW and SS formed 365 L/kgVSa of biogas, while a FW–SS ratio of 10:90 limited the biogas production to 293 L/kgVSa.
[114]	FW (24.0% TS; 78.0%TS VS); SS (0.7% TS; 70.0%TS VS)	Three scenarios were assessed through Life Cycle Assessment and Life Cycle Costing: (i) co-digestion of FW and SS; (ii) preliminary dark-fermentation of FW and SS, followed by a second step of anaerobic digestion; (iii) composting of FW and anaerobic digestion of SS. The functional unit was the annual amount of inlet waste (189,000 t/y of FW and 15,500 t/y of SS) to an actual Italian plant.	Co-digestion of SS-FW provided general environmental improvements with respect to separate SS anaerobic digestion and FW aerobic composting. The higher energy recovery due to the improved specific gas production of the digestion step significantly influenced the environmental credits. Despite that both the studied systems were economically sustainable, co-digestion has a shorter time of return of investment and a higher net present value than mono-digestion.

Table 5. Cont.

Reference	Substrates Tested	Process Parameters	Results
[115]	FW from MSW (25.7% TS; 87.9%TS VS); SS (5.5–6.6% TS; 85.5–88.1%TS VS)	Through Life Cycle Assessment, the current scenario (disposal of FW with MSW in landfill with energy recovery) and co-treatment (anaerobic co-digestion of source separated FW and SS) were assessed.	Co-digestion has less environmental impact for all categories modelled (except human toxicity) but needs a preliminary collection and pretreatment of MSW. Co-digestion has a 100% likelihood of a smaller global warming potential, and for acidification, eutrophication, and fossil fuel depletion, it carried a greater than 85% confidence of inducing a lesser impact than the current waste service.
[116]	FW and SS	Mono-digestions of SS and FW, respectively were compared to the co-digestion treatment from an environmental point of view.	The possible outcomes from co-digestion could be categorized as neutral, synergistic, and antagonist if methane production from SS-FW mixture is equivalent, higher, or lower than the sum of mono-digestion. When an antagonistic situation happens, the co-digestion system became much less favourable, although it required less water consumption and land footprint.
[117]	FW (40.3% TS; 88.6%TS VS); SS (7.5–25.0% TS; 70.0–75.0%TS VS)	Anaerobic co-digestion of SS and FW was compared to mono-digestion of FW and composting through a techno-economic analysis.	The introduction of SS in anaerobic digestion increased costs and payback time, rather than generating a higher waste amount and lower biogas yield. However, both the anaerobic digestion solutions resulted in general advantages over composting.
[118]	FW and dewatered SS at 3:10 mixing ratio	Incineration, anaerobic digestion, and co-digestion of SS and FW were evaluated through Life Cycle Assessment.	Anaerobic digestion and co-digestion resulted in low environmental impacts and high energy production. With their combination, the performance in terms of human health, ecosystems, and energy production improved by 36, 13, and 61%, respectively, compared with incineration (almost two times better). The performance remained the highest even with an increase in SS.
[119]	Diluted FW (1.2% TS; 92.1%TS VS; 4.7 pH); Primary SS (3.1% TS; 64.7%TS VS; 5.9 pH); Secondary SS (1.0% TS; 73.9%TS VS; 6.8 pH)	Lab-scale test of three different co-digestion mixtures (FW + primary SS; FW + secondary SS; FW + primary SS + secondary SS) at three different mix ratios (1:3; 1:1; 2:1; volume basis).	All feedstocks' combinations showed enhanced H <sub>2</sub> production as compared with the individual waste. A SS:FW ratio of 1:1 was found to be the best among the ratios tested, achieving a H <sub>2</sub> yield of 112 mL/g VS.

VSa: vs. added; VSr: vs. removed.

**Table 6.** Literature review of combined anaerobic digestion and composting process of FW and SS in terms of substrates tested, process parameters, and results achieved.

Reference	Substrates tested	Process parameters	Results
[120]	FW mixed with two anaerobically digested sludges working under mesophilic (37 °C) and thermophilic (55 °C) conditions to get a TS content of 20%.	Full-scale anaerobic digestion of SS and FW at different process parameters (37.4–54.8 °C; 6.9–13.5 kgVS/m <sup>3</sup> d; 7.8–14.7 d) was carried out. A further pile-composting was performed on the digestate, mixed with bulking agent (35%–65% mixing ratio)	The pile temperature during composting from thermophilic effluent never exceeded 65 °C, while short peaks over 70 °C were reached in the pile with mesophilic effluent because of the poor efficacy of heat removal throughout. For the same reason, the latter pile resulted in a higher temperature variability. Thermophilic effluent affects the composting step more positively than the mesophilic outlet sludge since it showed a faster disappearance of phytotoxicity and colonization by nitrifying. Composting efficaciously hygienized both starting mixtures since the absence of Salmonella and similar microbial communities were found in both final composts. The water initially introduced in the system was completely absorbed.
[121]	FW and dewatered SS at 1:2 mixing ratio	The environmental profile of anaerobic co-digestion of FW and SS in small plants with and without aerobic composting post-treatment was assessed.	Both anaerobic co-digestion of SS and FW with or without composting post-treatment permit a short supply chain that reduces the environmental impact due to transport, low energy requirement for the process itself, energy saving from the CHP unit, and energy/resources saving from the compost production. Global warming, acidification, eutrophication, and photochemical ozone creation from combined treatment were reduced by 69%, 54%, 61%, and 49% compared to those from anaerobic co-digestion only, respectively, while ozone depletion improved by 33%. Also, social advantages may be gained, such as increased public acceptability of waste treatment facilities and increased awareness among citizen on waste management issue.
[122]	Mixture of FW and SS at 5–6% VS	Two reactors operating under anaerobic (37 °C; 7 L volume; 30 d retention time; 1.6 gVS/L d) and aerobic (2.1 L; 12 d retention time; 0.5 gVS/L d) conditions, respectively. FW-to-SS mixing ratio of 0.42, 0.83 and 0.53 were adopted.	The optimum loading rate was found to be 1.6 mg VS/L d, resulting in a stable operation of the anaerobic compartment. A loading rate of 2.0 mg VS/L d increased ammonia concentration for the gradual accumulation of VFA, with acute stability loss and performance deterioration. Optimum performance of the aerobic system was achieved with an oxygen concentration of 4 mg/L, which resulted in 74% conversion of ammonia nitrogen. Under optimum conditions, the combined system yielded total removal rates of 93% vs. and 94% COD, with a high specific methane yield of 845 L/kg vs. and a CO <sub>2</sub> -to-CH <sub>4</sub> ratio of 0.63.

#### 4.2. Anaerobic Digestion

In general, AD involves breaking down organic matter in the absence of oxygen, producing biogas (primarily methane and carbon dioxide) and digestate (a nutrient-rich residue). The AD process mechanism is composed of four main stages: (i) hydrolysis, which consists of an enzymatic breakdown of complex organic polymers (carbohydrates, proteins, and fats) into simpler monomers (sugars, amino acids, and fatty acids); (ii) acidogenesis, where monomers are converted into volatile fatty acids (VFAs), alcohols, hydrogen, and carbon dioxide by acidogenic bacteria; (iii) acetogenesis, consisting in a further breakdown of VFAs into acetic acid, hydrogen, and carbon dioxide by acetogenic bacteria; (iv) methanogenesis, where acetic acid, hydrogen, and carbon dioxide are converted into methane and water by methanogenic archaea. Anaerobic co-digestion of FW and SS is an advanced waste management process that combines these two types of organic waste in a single anaerobic digestion system.

Since FW has low pH buffering capacity while the values for SS are relatively higher, co-digestion of FW and SS is a suitable option to balance the pH, resulting in a more stable and efficient process [119]. System stability was improved in co-digestion systems with co-substrate acting as a diluting agent to toxic chemicals like ammonia or  $\text{Na}^+$  as well as enhancement for biogas production [103]. The synergistic enhancements in terms of improved process stability or biogas generation are helped by combining the distinctive properties of the co-substrates with respect to trace elements, buffering capacity, and high easily biodegradable components. Specifically, in a co-digestion system, SS contributes to the more stable process and healthy microbial community due to its abundant trace metals. In contrast, FW ensures the more economic viability of the process due to the high biodegradable organics [111]. For these reasons, co-digestion of SS and FW is characterized by a more stable process, an abundant microbial community, and a higher methane yield. Such stability, as well as improved biogas generation, is confirmed even if the FW is pre-fermented due to the storage of such waste during its collection. Indeed, pilot-scale tests have revealed how a digester fed with SS and pre-fermented FW achieved faster pH recovery (due to the addition of FW) and relatively stable biogas production throughout the operation [106].

A SS/FW ratio of 1:1 (VS basis) achieved a significant methane yield (799 mL/g VS), which was much higher than that obtained during mono-digestion. The synergistic effect was also confirmed by the high VS and COD removals, while pH and intermediate inhibitions (e.g., volatile fatty acids) were not observed [108]. Similar results were achieved by Pan et al. [109], where, at the same SS-FW mixing ratio, not only methane yield was maximized but also the lag phase was shortened. In addition, co-digestion of SS with FW was able to mediate the  $\text{N}_2\text{O}$  emissions while simultaneously improving the digestate dewaterability [109]. The adoption of thermophilic (55 °C) and mesophilic (35 °C) temperatures in SS-FW co-digestion could increase the methane yield obtained from FW mono-digestion up to 85.3% and 35.3%, respectively [112]. On the other hand, a predominance of SS in the mixture (90%) could decrease the methane production to 293 mL/g VS [113]. Thakur et al. [105] showed how adding 2% of FW to the pilot-scale SS AD can overcome the need for adding additional micro-nutrients (such as iron, cobalt, nickel, and zinc) which are common in mono-digestion, requiring much less effort for maintaining the optimum pH range inside the reactor. This feeding strategy can be feasible for integrating an anaerobic digester in a decentralized domestic wastewater treatment plant.

However, an optimal FW concentration of 33.3% could be found in the literature due to the variation in SS and FW properties [33]. The adoption of a co-digestion strategy at the suggested mixing ratio could produce about  $5.83 \times 10^9 \text{ m}^3$  of methane (equivalent to 22.86 billion kWh electric energy) annually, which is more than enough to meet the annual electric energy demand in China's wastewater treatment plants [110]. Under proper pH conditions, co-digestion of SS and FW could be used also for hydrogen production: an SS:FW ratio of 1:1 was found to be the best combination in terms of process stability and hydrogen yield [119].

The methane production of FW–SS co-AD can be further enhanced by recirculating the biogas produced in the reactor. Indeed, the synergistic effect of CO<sub>2</sub> acidification and high VFA production led to the benefit of reduced digester pH (from 8.3 to 6.6), NH<sub>3</sub> control (2380 mg/L), and in situ CH<sub>4</sub> enrichment (88%) [104]. An optimum intermittent biogas recirculation could be used as an alternative mixing method for the large-scale AD, for enhanced biogas yield, reduced impeller power consumption, and improved energy efficiency.

Mattioli et al. [123] studied the effects of co-digestion of separately collected organic fractions of municipal solid waste and mixed sludge in an industrial wastewater treatment plant. In terms of mass balance, the co-treatment of 4430 kgTS/d of thickened SS and 2624 kg/S/d of pretreated FW resulted in a dewatered sludge of 3013 kgTS/d and 2629 m<sup>3</sup>/d of biogas. The co-digestion allowed a substantial increase in biogas generation compared to the SS mono-digestion, resulting in a clear increase in energy recovery. Despite the energy demand increasing from 7771 kWh/d (SS mono-digestion) to 9530 kWh/d (co-digestion), potential power generation rose from 3900 kWh/d to 7800 kWh/d, which is 85% of the total energy consumption of the wastewater treatment plant.

The increased loadings from the addition of FW do not deteriorate digester performance, but, on the contrary, improve the digestibility of SS. [124] In addition, co-digestion of FW and SS can save about 110 kWh/FU of energy [125].

Co-treatment of FW and urban wastewater in wastewater treatment plants could be a plausible management strategy. However, the increased organic load in the wastewater influent would impact the capital and operating costs of the WWTP, mainly due to the increase in sludge production [126]. On the other hand, the co-treatment of FW and wastewater would be more environmentally friendly than their separate treatment, but its economic feasibility strongly depends on the ratio between the management costs of FW and SS [127].

In general, co-fermentation processes of FW and SS are economically sustainable, as they report positive net present values in 20 years (up to EUR 10,518,291). Biogas upgrading-based scenarios are characterized by higher net present values; nevertheless, the combined production of heat, electricity, and biomethane is the most cost-effective option, thanks to biomethane revenues and electricity sales. This means that CHP should not be neglected, as the optimal configuration may lie in the combined recovery of biomethane, electricity, and heat [128]. From an environmental point of view, co-digestion of SS and FW provides general environmental improvements compared to SS mono-digestion and SS-FW composting, mainly due to the higher energy recovery during the anaerobic digestion step [114]. Anaerobic co-digestion of FW and SS could reduce 90% of the greenhouse gas emissions and 83% of the energy consumption of an integrated treatment system in Hong Kong [129]. Similar achievements could be extended also to general MSW: despite the necessity of waste collection and pre-treatment, anaerobic co-digestion results in several environmental benefits [115]. On the other hand, if methane production from the co-digestion process is lower than the total production from the mono-digestion processes, this operation is much less favourable than mono-digestion [116]. Indeed, specific assessments on the actual methane yield from single and mixed substrates, as well as reactor configuration and optimized conditions, are necessary for a suitable and convenient adoption of large-scale co-digestion [8]. Environmental benefits could also be obtained by combining anaerobic co-digestion with other mono-digestion or incineration treatments. Indeed, Chiu et al. [118] reported how a combination of these treatments improved the whole waste management system performance in terms of human health, ecosystems, and energy production by 36, 13, and 61%, respectively, compared with incineration only. However, not all environmental benefits from FW co-treatment with SS can be readily monetized into revenue to support these projects. Indeed, important issues from inert impurities in FW have a significant implication to the planning, design, and operation of co-digestion plants [130].



The encouraging performance of FW–SS laboratory-scale co-digestion did not lead to the large-scale adoption of this option. At present, the leader in this approach is represented by the European Union (EU), where only 11% of the current AD plants for the FW treatment work in co-digestion with SS [131]. In particular, Germany and Italy are the most representative countries working in co-digestion systems, counting several operative plants [9]. Indeed, despite from an economic point of view, co-digestion systems appear comparable or slightly less favourable than mono-digestion ones; the economic performances are strongly higher than those of composting [117]. Indeed, as shown by the case of Hong Kong, implementing an anaerobic co-treatment of 380 t/d of FW and 240,000 m<sup>3</sup>/d of SS can save around HKD 747,668 compared to the current system [129].

Alternatively, co-digestion can be extended also to animal manure, which is regularly used in numerous businesses for methane biogas production. Though co-digestion can offer an increase in methane generation through a straightforward and portable process, transporting animal dung to the AD facility is tricky because animal manures are more dangerous to human health [132].

Table 5 summarizes several works dealing with the topic of FW–SS co-digestion, highlighting the substrates tested, the process parameters, and the main results achieved.

#### 4.3. Combined Anaerobic Digestion and Composting

Combining AD and composting results in achieving a biogas generation from the AD stage and producing compost from the further composting step. The combination of two biological co-treatments not only produces biogas, but reduces the digestate disadvantages (e.g., odour, high humidity and phytotoxicity, pathogens etc.) prior to its agricultural use as compost.

Combined anaerobic digestion and composting of an FW–SS mixture is rarely reported in the literature and only few aspects can be pointed out. Little information is currently available on the composting of anaerobic co-digestates and on the assessment of the process at both the laboratory and industrial levels. However, all the advantages from single co-composting and co-digestion treatments could be extended to the combined treatment. The main advantages can be identified as better solids reduction, improved sludge dewatering properties, and reduction of nitrogen [133].

Optimum conditions of the FW–SS anaerobic co-digestion process followed by aerobic composting of the digestate were identified by Ghanimeh et al. [122]. At a loading rate of 1.6 mg VS/L d and FW-to-SS mixing ratio of 0.83, average removal rates of 84% ± 4% TS, 88% ± 1% VS, and 88 ± 2% COD were achieved, as well as specific biogas and methane yields of 1548 L/kg vs. and 843 L/kg VS, respectively. A loading rate of 2.0 mg VS/L d and a prevalence of SS in the mixture increased ammonia concentration due to the gradual accumulation of VFA, with acute stability loss and performance deterioration. Under optimal conditions, the aerobic compartment achieved an oxygen concentration of 4 mg/L, which resulted in 74% conversion of ammonia nitrogen. Thus, the overall system yielded high (>90%) total removal rates of vs. and COD, with a high specific methane yield [122].

The best conditions from a combined anaerobic co-digestion of FW and SS and composting process could also be achieved with a retention time of 8 days during the co-digestion stage and the adoption of thermophilic SS, since it ensures a faster reduction of digestate phytotoxicity and increase in nitrifying bacteria [120].

The integrated anaerobic/aerobic co-treatment process presents definite economic advantages with respect to the single aerobic process. The savings mainly come from the reduction of the composting unit size due to the pre-digestion of a fraction of the substrate [120].

The adoption of composting post-treatment to anaerobic co-digestion of SS and FW may offer an environmentally and social sustainable option of waste than anaerobic co-digestion only [121]. Global warming, acidification, eutrophication, and photochemical ozone creation from combined treatment were reduced by 69%, 54%, 61%, and 49% compared to the ones from anaerobic co-digestion only, respectively. While ozone depletion resulted in an improvement of 33% [121]. In addition, the economic value of the compost produced from this process ranges between 24.7 to 61.6 EUR/ton, having a better marketable value when the total content of P is higher, since this element represents 70% to 80% of the total value [134].

Table 6 summarizes several works dealing with the topic of FW–SS combined co-composting and co-digestion, highlighting the substrates tested, the process parameters, and the main results achieved.

## 5. Thermal Co-Treatment of FW and SS

Thermochemical treatment of SS and FW has gained increasing attention owing to its advantages of nutrient and energy recovery, especially hydrothermal treatment and pyrolysis technologies, from an environmental protection and economic perspective. On the other hand, treatments strictly dependent on the water content of the inlet waste, incineration and gasification, are not identified as the most suitable solutions for the co-treatment of SS and FW, substrates with elevated humidity levels. Indeed, the co-treatment with SS is usually done using woody biomass [135], forestry waste [136], or energetic crops [137]. Indeed, for the case of incineration and gasification, environmental analysis also reveals how these technologies are less preferred for the co-treatment of SS and FW than other ones, such as conventional biological processes [118] or advanced thermal processes (pyrolysis or carbonization) [138]. Furthermore, full-scale application is still scarce in several developed countries and most of them are carried out only at the pilot scale.

For all these reasons, this section discusses the main thermal technologies that have achieved a viable performance for the co-treatment of FW and SS: pyrolysis and HTC.

### 5.1. Pyrolysis

Pyrolysis starts with an initial loss of moisture from the feedstock, followed by the thermal decomposition phase, which is a breakdown of complex organic polymers into smaller molecules (depolymerization) and a further breakdown of large hydrocarbon molecules into smaller hydrocarbons (cracking). In the end, volatile compounds are condensed to produce bio-oil while the remaining solid materials are carbonized into biochar. The biochar can be used for nutrient recovery, but the main resource utilization consists of energy recovery, using bio-oil (directly or refined), syngas for producing transportation fuels, heating oil, or chemical feedstocks.

Co-pyrolysis is a cost-effective way to improve the quality of pyrolysis products without changing the conventional production process, while taking the characteristics of SS and FW into account [139].

Co-pyrolysis of SS with heavy metal-free organic waste (e.g., FW) could improve both the yield and quality of gas, bio-oil, and char, which is due to the catalysis of alkali metals in SS during the co-pyrolysis process [140–142]. The combined process can increase the C content, reduce the ash content, create well-developed pore structure, and dilute heavy metal concentrations of blended biochar [143–145].

Compared with SS pyrolysis, co-pyrolysis of SS and various organic waste leads to lower biochar yields but with carbon contents in blended biochars. Co-pyrolysis further promotes the transformation of heavy metals in the biochar into more stable forms, reducing their associated environmental risk [146]. The ratio of 70:30 for SS–FW co-pyrolysis is the best since it has a synergistic effect on the pyrolysis in different temperature ranges and can also greatly suppress CO<sub>2</sub> emission, maintaining a high value of CH product gases [147].

The benefit of pyrolysis can be extended also for the output from anaerobic co-digestion of SS and FW. Indeed, co-pyrolysis of SS and FW digestate increased the biochar aromaticity and pH, decreased its phytotoxicity, and reduced the contents and instability of both total and bioavailable heavy metals, suggesting how high levels of mineral components in the digestate can immobilize more heavy metals in biochar [148]. In addition, from an environmental point of view, pyrolysis of dewatered sludge from anaerobic co-digestion of SS and FW generates the highest environmental performance in terms of GHG emission and P recovery from ash, among other thermal and biological single and co-treatments for SS and FW [138]. Co-pyrolysis is an eco-friendly SS treatment method since it can reduce the overall environmental burden by 58–83% (especially for global warming potential and terrestrial ecotoxicity), achieving a net positive energy balance as well [149]. The effect on the environment of nitrogen-related pollution aroused by this method should be noted. Excess nitrogen in the atmosphere brings about a range of environmental issues, including ocean acidification, climate change, eutrophication of water bodies, ozone depletion, groundwater pollution, and toxic ground-level ozone pollution, which constitutes a serious risk to the natural environment and human beings [150].

However, the synergistic effects of SS and FW in co-pyrolysis in terms of biochar enrichment and heavy metals immobilization are not clear. Therefore, the study of the co-pyrolysis of SS and FW and its upgrading at full scale has practical significance and can provide a new direction for the treatment of the two waste streams. However, the use of a pyrolysis temperature of 550 °C for organic waste mixtures avoids the volatilization of Cd into the bio-oil and gases (which further leads to secondary pollution due to its low volatile temperature), which occurs at a processing temperature of 750 °C [151].

In economic terms, pyrolysis is characterized by high capital (17–25 USD/ton) and operational (2–3 USD/ton) costs, as well as problems related to licensing for commercial applications of its products (e.g., employing biochar as a fertilizer or soil conditioner) [152]. In general, this technology is financially superior to incineration-based technology based on the data of net present value, benefit-cost ratio, and internal rate of return [153]. However, pyrolysis is characterized by capital costs of the reactor, combustor, cyclone, compressor and pump, storage, and supplementary equipment, and operating costs consisting of feedstock, silica sand for non-catalytic pyrolysis or catalyst for catalytic pyrolysis, energy, labour, and maintenance costs [154].

For the scalability of this process, it is necessary to design an optimized pyrolysis system based on the composition, topography, economic viability, amount of waste, and weather conditions for the most efficient recovery of energy, bio-oils and bio-char. Nevertheless, the scientific literature about multiple parallel reaction models is still limited, and more kinetics based on these models should be developed [155]. In addition, pretreatment techniques are essential to achieving a high-grade pyrolysis bio-oil due to the complex nature and high moisture content of both SS and FW [156]. Moreover, drying and grinding must be performed before pyrolysis to obtain small particle sizes, which improve the reaction efficiency and bio-oil yield. While pretreatment is an energy-intensive process, energy costs will be huge during the process [157].

Table 7 summarizes several works dealing with the topic of FW–SS co-pyrolysis, highlighting the substrates tested, the process parameters, and the main results achieved.

## 5.2. HTC

HTC occurs in a water-saturated environment under high temperatures (180–250 °C) and pressures (up to 4 MPa), which facilitate the breakdown of organic matter into hydrochar, gases, and liquid by-products. Water acts as a reactant, solvent, and catalyst, facilitating hydrolysis, dehydration, decarboxylation, and aromatization reactions. Reacting with water, a breakdown of complex organic compounds (carbohydrates, proteins, and lipids) into simpler molecules occurs (hydrolysis). Then, water molecules are removed from organic compounds, leading to the formation of more stable structures like furan derivatives (dehydration). Further, carbon dioxide (CO<sub>2</sub>) is released from organic acids, contributing

to the reduction of oxygen content in the solid product (decarboxylation). Finally, the aromatization takes place, in which aromatic structures are formed, enhancing the stability and carbon content of the hydrochar [158]. The HTC process dynamics consist of an initial heating of the feedstock to the desired temperature and pressure, initiating hydrolysis and decomposition of organic matter. These conditions are maintained until the hydrochar and by-products are completed, and then there is a gradual cooling of the reactor, leading to the precipitation of hydrochar and separation of liquid and gaseous by-products. The whole HTC process lasts from 1 to 8 h: the longer the duration, the higher the carbon content and energy density of the hydrochar. The main resource recovery of HTC is nutrient recovery through the utilization of hydrochar or process water (containing numerous nutrients such as nitrogen and phosphorus) for solid and liquid fertilizer, respectively. Furthermore, HTC allows for energy recovery, using the hydrochar as fuel (directly or further processed into activated carbon).

The hydrochar produced from SS is usually of high ash content (i.e., 75%–85%) and a high heating value (HHV), strictly dependent on the moisture content (from about 8 MJ/kg when humidity is 97% to almost 20 MJ/kg when the humidity is less than 70%), which largely limits its utilization as a value-added solid fuel [159,160]. To improve the fuel properties of SS-based hydrochar, co-HTC of SS with solid waste containing a high organic matter content such as FW could be conducted [161].

SS and FW have been proven to have a synergic effect on the HTC process, with better results than the single HTC process of SS or FW [162]. Hydrochars derived from co-HTC of SS and FW possess better comprehensive combustion characteristic indexes than those obtained from SS HTC, such as higher C and HHV content, and lower N, S, and O, content [163]. All these benefits are not significantly affected by mixing ratios, but the chemical composition of additives plays a significant role in the synergy effect of SS and FW [164]. For example, co-HTC of sludge and cellulose- or lignin-based FW (e.g. vegetables and fruits) produce clean C-rich sludge-based hydrochar for thermal utilization [165]. Indeed, compared to the outputs from SS HTC, the outputs from co-HTC of SS and FW generate higher-quality fuel with good combustion performance (HHV of 6.86–12.90 MJ/kg), reduce heavy metals (Ni, Cr, and Cu), and are safer and have a lower potential ecological risk index [166]. In addition, the co-HTC of SS and starch-based FW (i.e., rice and noodles) produces N-rich hydrochar for soil amendment material and adsorbents (36.8–50.9% higher than those produced from HTC of SS) [165]. Fertilization with both processed water and biochar from the HTC of 60% SS and 40% FW mixture (enriched by chemical nitrogen at 15 kg da<sup>-1</sup>) significantly improved plant height, plant fresh and dry weight, and root dry weight parameters of garden cress over the previous treatments [167].

The use of the hydrothermal process at 140 °C for a FW and SS mixture as a pre-treatment of co-digestion increases the biogas production by 50% compared to the untreated one. Indeed, the hydrothermal process modifies the physical properties of the FW–SS mixture, enhancing the solubilisation of organic solids [168]. In terms of mass, the HTC of a 1 kg/h FW-SS mixture stream can produce from 71% to 77% hydrochar and about 10% gases, according to the composition of the mixture [169]. In terms of energy, HTC has an energy consumption of about 1.6 MJ/kg, which can be balanced with an energetic output of about 2 MJ/kg; however, the input and output of energy and the recovery rate are strictly dependent on the mixture of waste in the input [170].

**Table 7.** Literature review of co-pyrolysis of FW and SS in terms of substrates tested, process parameters, and results achieved.

Reference	Substrates Tested	Process Parameters	Results
[146]	SS (49% ash; 6.85 pH) FW in different forms (bamboo sawdust, wood sawdust, rice husk, exhausted tea, and kitchen waste).	Single pyrolysis of SS was conducted at 350, 550, or 750 °C. Co-pyrolysis of SS and FW (mixing ratio of 4:1 SS:FW) was conducted at 550 °C for 60 min.	A temperature of 550 °C was the most effective in consolidating the heavy metals into the biochars and preventing the Cd transforming into gaseous phases. Co-pyrolysis of SS and FW led to lower biochar yields but with higher pH values (increased between 21.80% and 31.70%) and carbon contents (raised between 33.45% and 48.22%) in blended biochars. Co-pyrolysis promoted the transformation of heavy metals in biochar into more stable forms, which significantly reduced their associated environmental risk.
[147]	Dried SS. FW composed of cabbage leaves (43%), orange peel (17%), rice (26%), and pork (14%).	FW-SS mixtures (0, 10, 30, 50, 70, and 100% w/w) were heated from 30 to 900 °C at three heating rates (10, 20, and 40 °C/min). The change of gas functional groups was detected by FTIR.	The ratio 30% SS and 70% FW had a synergistic effect on the pyrolysis in all temperature ranges and could also greatly suppress CO <sub>2</sub> emission (−35.25%). At that ratio, the gas products were mainly nitrides, hydrocarbons, and furans.
[148]	SS (55.05% ash; 6.85 pH) FW digestate (51.92% ash; 7.34 pH)	Co-pyrolysis of SS and FW digestate at different mixing ratios (4:0, 3:1, 2:2, 1:3, and 0:4; SS:FW digestate w/w) at 550 °C for 60 min.	Co-pyrolysis increased the aromaticity and pH (by 13.22–26.56%) of the blended biochar, and significantly reduced the contents of total and bioavailable heavy metals. A transformation of Cr from the residual fraction (F4) to the oxidizable fraction (F3) was possible when the FW digestate/SS ratio was ≥3:1. Heavy metal-associated ecological risk (potential ecological risk index lower than 15.51) and phytotoxicity (germination index higher than 139.41%) of the blended biochar were reduced in co-pyrolysis. High levels of mineral components in FW digestate greatly immobilized more heavy metals in biochar.
[138]	SS (45 g/capita*d) and FW (250 g/capita*d) per 10,000 habitants.	LCA was applied to identifying the best environmental profile of several alternative technologies (low-temperature incineration, composting, cement feedstock, low-temperature carbonization, dry granulation, pyrolysis, and high-temperature incineration) by transferring the disposal of FW from a waste incineration plant to an SS treatment plant.	For each of the technologies compared, the combined SS–FW digestion system was superior to the separate ones. Among the various technology configurations of the combined system, pyrolysis and composting were judged superior with respect to GHG emissions and P recovery. When a detailed consideration of health risks was added to the analysis, pyrolysis was identified as the best option to target for technology renewal because of its superior disability-adjusted life year to that of composting, which caused greater heavy metal emissions.



Challenges must be faced to spread this technology on a commercial scale, and the development of reliable predictive models is considered by the research community as one of these challenges [171]. The higher expenses within the HTC process cannot be compensated for by additional energy production and agricultural yields. However, the optimization of the HTC process chain through integrated SS digestion and process water recirculation leads to significant reductions in GHG emissions of HTC [172]. In general, mixing SS and FW generates environmental benefits regardless of the mixture composition if the residues are used as raw materials. In particular, the main benefits involve the environmental impact categories of non-carcinogen toxicity, global warming, and ecotoxicity on freshwater and marine environments [173]. From an energetic point of view, the potential energy of the solid product (hydrochar) is almost equal to the input energy. This suggests that HTC can achieve a balance of input and output energy for processing SS with almost no additional energy supply. However, other HTC schemes can achieve a higher efficiency compared with traditional HTC (a detailed energetic balance can be found in ref. [174]).

The economic performance of HTC was studied by Zeymer et al. [175], which found that the minimum cost of hydrochar was 151 EUR/ton in a medium-sized unit (500 kW). This shows that the price of HTC is competitive with the price of conventional SS treatments (160 EUR/ton for agricultural use and 330 EUR/ton for combustion) [176]. The cost of the HTC treatment process can be reduced by producing steam and electricity using waste heat recovery. Furthermore, the expansion of the scale of HTC, the potential improvement of hydrochar yield, and the support of public policies can enhance the overall economic potential of HTC [177]. It should be noted that the current economic analysis is mainly based on the data from laboratory-scale experiments, and continuing scientific exploration of HTC is crucial.

The possible extensive and industrial application of HTC depends on the characteristics of the treated waste. HTC of pre-treated waste shows an almost constant solid mass yield trend and a moderate increase in gas production with the increase in process temperature. HTC of wet samples leads to a significant and rapid increase in gas production with reaction temperature when compared to corresponding pre-treated samples [178].

In addition, it is important to mention the issue related to the effluents derived from the HTC process. For example, in HTC per dry ton of hydrochar is produced approximately 2 m<sup>3</sup> of effluent that needs to be treated at a cost [179]. These effluents are acidic, with high electrical conductivity (according to the amount of electrolytes in the feedstock) and highly contaminated with organic matter; however, BOD<sub>5</sub> to COD values indicate good biodegradability of the effluent [180]. Due to their characteristics, the effluents from HTC are critical for the economic and ecologic viability of the HTC technology.

Table 8 summarizes several works dealing with the topic of FW-SS co-HTC, highlighting the substrates tested, the process parameters, and the main results achieved.

**Table 8.** Literature review of HTC of FW and SS in terms of substrates tested, process parameters, and results achieved.

Reference	Substrates Tested	Process Parameters	Results
[162]	Mixture of dried, crushed, and sieved SS and FW as banana stalk at SS concentration of 0%, 30%, 50%, 70%, and 100%.	HTC process at different temperatures (180 °C, 230 °C, and 280 °C) for 60 min. Hydrochar was dried at 105 °C for 24 h, crushed, and then sieved at 178 µm.	The synergistic effects of SS and FW were mainly induced by Maillard and Mannich reactions. Hydrochar resulted in superior yield, C and N contents, higher heating value, and energy yield, with the optimal increase rates reaching 12.96%, 6.02%, 142.97%, 10.85%, and 22.16%, respectively. Co-HTC allowed the redistribution of the speciation of heavy metals, which migrated from direct/potential toxic (F1 + F2 + F3) fractions to non-toxic (F4) fraction, thus greatly reducing the contamination degree of heavy metals in hydrochar (except for Pb).
[163]	Mixture of dried, crushed, and sieved SS (41.16% ash) and FW (6.14% ash) at SS concentration of 0%, 30%, 50%, 70%, and 100%.	HTC process at different temperatures (180 °C, 230 °C, and 280 °C) for 60 min. Hydrochar was dried at 105 °C for 24 h, crushed and then sieved.	Co-HTC resulted in an improvement in C content, HHV, and fuel ratio of the hydrochar. The hydrochar obtained at 230 °C with a SS content of 30% possessed the highest HHV of 22.87 MJ/kg and fuel ratio of 0.36. At these conditions, both ignition and burnout temperatures declined, indicating the enhancement of thermal characteristics.
[165]	SS (13.5% TS) and FW model compounds as microcrystalline cellulose, xylan, lignin, and starch	HTC process at 220 °C and 4.5 MPa for 30 min. The gas products were collected in a gas bag and then bubbled into H <sub>2</sub> SO <sub>4</sub> (0.1 M) solutions to collect the N-containing gas.	Most of the nitrogen in the SS was transformed into organic-N (44.6%) and NH <sub>4</sub> <sup>+</sup> (23.3%) in the aqueous product, and only 20.3% of nitrogen was retained in the hydrochar. The added model compounds could react with N compounds in aqueous products through Maillard and Mannich reactions, leading to an increase in the retention rate of nitrogen to 36.8–50.9%, especially upon the addition of starch and xylan.
[166]	Mixture of dried, crushed, and sieved SS (50%) and FW model compounds such as xylan, lignin, and cellulose (50%).	HTC process at 220 °C for 60 min. Liquid and solid phases were separated by vacuum filtration through a filter paper (0.4 µm). Then the hydrochar was at 105 °C for 24 h.	The majority of Cr, Ni, Cu, and Zn was still accumulated in hydrochar during individual HTC of SS. The addition of lingo-cellulosic biomass could effectively reduce (F1 + F2) fractions of Ni, Cu, and Cr, exhibiting the lowest potential ecological risk index. The fuel ratio of hydrochar derived from co-HTC increased to 0.08–0.39 and high HHV (6.86–12.90 MJ/kg) was also achieved. The combustion behaviors of hydrochars derived from co-HTC were expected to be safer and stable than that of hydrochar derived from SS.

Table 8. Cont.

Reference	Substrates Tested	Process Parameters	Results
[167]	Mixture of 60% SS and 40% FW.	First depolymerisation at 150 °C under 5–8 bar pressure followed by hydrolysis at 250 °C and 50 bar.	Fertilization of soil with HTC bioproducts at the highest nitrogen dose significantly improved plant height, plant fresh and dry weight, and root dry weight parameters of garden cress. The vitamin C content in cress decreased with increasing levels of nitrogen. HTC bioproduct fertilization applications improved chlorophyll a, b, and the total contents of garden cress leaves. Moreover, the nitrate (NO <sub>3</sub> ) concentration of cress increased with CN doses while it decreased in all BC and PW administrations. Plant nutrient content was positively affected in all fertilization applications, except for Na and Cl.
[168]	FW (15% TS; 83%TS VS; 4.27 pH); SS (10% TS; 70%TS VS; 6.12 pH)	HTC process of 200 mL SS–FF mixture (1:1) in a 500 mL reactor at different temperatures (from 80 to 180 °C) and pressures (from 0.8 to 1.8 MPa) for 30 min.	Co-HTC was shown to increase the COD and VFAs content of the inlet waste for the following anaerobic digestion step. A maximum increase in biogas production of 50% is achieved with an HTP temperature of 140 °C.
[173]	FW (16.8% TS) SS (24.6% TS)	Environmental performance of three different blends (1 kg as functional units) processed through an HTC process at 200 °C for 1 h.	All blends generate environmental benefits, especially for the environmental impact categories of non-carcinogen toxicity, global warming, and ecotoxicity on freshwater and marine environments. Blend 1 (42.0% SS, 54.7% FW, and 3.3% Pruning) has lower environmental impacts, compared to blends 2 and 3, since blend 1 has a higher mass yield, calorific value, and moisture.

## 6. Conclusions and Future Perspectives

The aim of this work was to review the status of current biological and innovative thermal co-treatments of FW and SS, focusing on the technical feasibility and the environmental benefit of these solutions.

The key message of this work is that, concerning conventional biological process, both aerobic and anaerobic co-treatment of FW and SS can contribute positively to the process in technical and environmental terms. Indeed, the synergy of the two substrates is evident in composting and especially anaerobic digestion. The optimum solution could be achieved by anaerobic co-digestion followed by composting, which combines the benefits from each process, though further investigation should be carried out. On the other hand, the waste treatment plant and the process parameters need to be readapted if the co-treatment is pursued.

Regarding the thermal process, co-treatments of FW and SS such as incineration and gasification have limited application at both full and pilot scale, due to the unsuitability of the substrates related to their high humidity and requirement for an expensive drying process. On the other hand, interest is growing around pyrolysis and hydrothermal carbonization, which appears to be less dependent on the level of moisture of these substrates. Co-pyrolysis and hydrothermal carbonization can generate an output with enhanced thermal properties such as heating value or carbon content for energy use. Furthermore, the outputs from these processes could be applied also as fertilizer, thanks to the synergic effect between FW and SS in reducing the level and the availability of toxic compounds such as heavy metals.

By adopting the co-locating concept, which offers a new insight for the future of SS and FW management towards greater environmental and economic sustainability, future waste treatment plants can be transformed into powerful profit producers instead of consumers, according to the analysis of the engineering feasibility and economic viability.

In addition, this system could be upgraded easily to a biorefinery vision: the term ‘biorefinery’ is related to the priority of the extraction and production of valuable biomolecules rather than energy production. In this sense, the benefit resulting from the energy recovery could be increased by the production of high-added value bioproducts such as VFA and biopolymers.

The application of this co-treatment would depend on the level of socio-economic development of a country or region and other institutional and political constraints. The mix of different approaches and technologies could be the most promising and successful scenario.

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