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**Resource recovery from food waste via biological processes**

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**Abstract**

The need for sustainable development is driving a major focus shift in the biowaste treatment sector. While traditional practices such as landfilling are progressively being banned/penalised due to their obvious environmental impacts and prevention policies are being implemented, the potential of food waste (FW) as a resource is being increasingly recognised. The immense amounts of food waste produced worldwide – and its overall characteristics – make it a promising candidate for resource recovery if collected separately, a practice that is gaining popularity. This chapter aims at giving a general overview of the different processes that are being developed/implemented for resource recovery from FW, including energy, carbon, and

nutrients (*i.e.* N and P). The main options are critically assessed and compared. In addition, a special focus is given to potential state-of-the-art research/technological developments that could be applied in the future. Key challenges to be faced are also discussed, including legislative, social and technological aspects.

## **Keywords**

Solid waste; valorisation; biogas; biohydrogen; biomolecules; single-cell

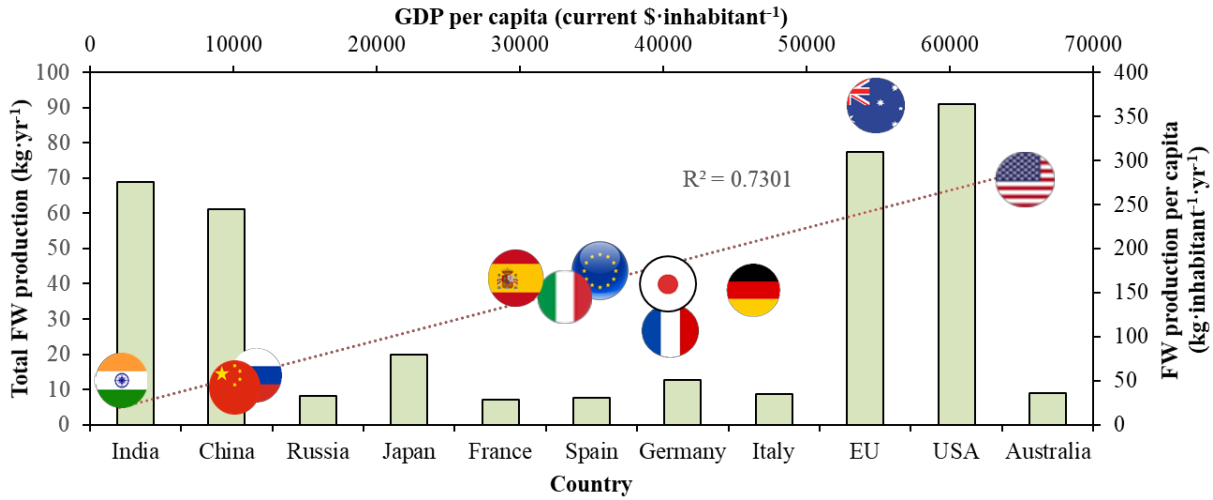
## **14.1. Food waste production and characteristics**

### **14.1.1. How much food waste is produced worldwide?**

FW can be precisely defined as the “mass of food lost or wasted in the part of food supply chains leading to edible products for human consumption” (Gustavsson et al., 2011). It is important to note that the FW definition shown above does not include the source-separated residues from industrial activities related to food production (*e.g.* dairy effluents, sugar-refining residues, brewery wastes or agricultural residues). As each of these industrial effluents has different characteristics and treatment/valorisation options, discussing them separately falls out of the scope of this chapter, which focuses on general, non-specific FW.

The costs of wasting food have always been in the minds of societies, either due to efficiency considerations (waste of resources), economic concerns (money losses), or due to its moral dimensions (considering simultaneous hunger and food wastage). In the last decades, humanity has also recognised the increasing importance of the environmental and social challenges associated to the increasing FW production, mostly related to population growth and globally increasing living standards. Although it is practically impossible to calculate precise numbers, it has been estimated that about 1.3 billion tons of food are lost along the food supply chain every year, which corresponds to one third of the total production for human food

consumption (FAO, 2012). This not only represents a missed opportunity for the economy and food security, but also a waste of all the natural resources used for growing, processing, packaging, transporting, and marketing food (FAO, 2015). The overall and per capita production of FW for different countries is shown in Figure 14.1, presenting also the relationship between the gross domestic product (GDP) per capita and the amount of FW produced. As population grows and living standards improve, the amount of FW is increasing tremendously, with projections stating that global FW production will be more than twice its current value by 2050 if consumption habits and food production practices do not change drastically (Huang et al., 2020). Therefore, if FW generation is to be reduced, wastage habits, particularly in high-income countries, must be modified and adapted to modern times. Prevention must be prioritised and, when not possible, re-use and recycling approaches must be favoured.



**Fig. 14.1.** Total food waste production per country (bars) and correlation between food waste production per capita and GDP per capita (flag position). Data from Statista (2019) and The World Bank (2019).

Opposed to general public perception, FW is not always mostly produced during the consumption step of the food chain. Food harvesting, processing, and distribution steps also represent significant fractions of the total FW generated (up to 50%, 25% and 20%; 95% altogether) (WWF, 2017). Nevertheless, this proportion changes between countries. As an example, while in sub-Saharan Africa consumer losses represent only the 5% of the food wasted, this number goes up to 52% in Europe and up to 61% in North America and Oceania (Jain et al., 2018), being affected by cultural and socioeconomic aspects. Therefore, reducing the amounts of FW produced per capita will not only require to change end-consumer habits (*e.g.* change consuming habits or avoid food spoilage at household level), but it will also demand systemic changes to motivate producers and retailers to reduce the amounts of FW associated to their activities. A recent example is the EU directive 2008/98/CE, which imposes the selective separation and organic valorisation of the FW from producers of more than 10 tons  $\text{FW}\cdot\text{yr}^{-1}$ . Applying more measures as such and penalising traditional waste management approaches (*e.g.* landfilling), is imperative to slow down (and eventually reduce) the current increase in FW production. An example at a household level is the pay-as-you-throw approaches, where users are charged a rate based on the amount of waste that they generate for collection. This strategy has been successfully implemented in several European, North American and Asian cities, showing an effective reduction in the amounts of waste produced (especially when combined with effective recycling and tax reduction to avoid extra costs to average users). Nowadays, it is just “too cheap” to generate and deal with waste.

From an environmental point of view, FAO (2015) reported that global food loss and waste generate annually 4.4  $\text{GtCO}_2$  eq., or about 8% of the total anthropogenic greenhouse gases (GHG) emissions. Thus, the contribution of food wastage emissions to global warming is close to the one caused by the global road transportation system (FAO, 2015). In fact, when comparing global GHG emissions due to food wastage with overall GHG emissions by country,

global food wastage would be the third largest emitting country in the world, after China and USA in absolute terms.

#### **14.1.2. FW characteristics: what resources can be recovered from FW?**

The potential and relevance of FW as substrate for resource recovery (*e.g.* carbon, nutrients or energy) lie mainly on: (i) the enormous amounts of FW being produced, (ii) its ubiquitous nature, and (iii) its appropriate physicochemical characteristics (Capson-Tojo et al., 2016; Dobbs et al., 2011). Regarding the latter, although the FW characteristics vary worldwide (mainly due to dietary habits), they are remarkably consistent. Table 14.1 shows average characteristics of FW from different countries.

**Table 14.1.** Food waste characteristics in different countries (adapted from Capson-Tojo et al. (2016)).

<b>Country</b>	<b>France</b>	<b>United Kingdom</b>	<b>Italy</b>	<b>China</b>	<b>USA</b>
<b>FW source</b>	Mixed source-sorted wastes	Domestic FW	Household	University restaurant	Waste management company
<b>Total solids (TS; %)<sup>1</sup></b>	21.0 ± 14.9	23.7 ± 0.1	27.5 ± 0.1	21.8 ± 0.3	30.9 ± 0.1
<b>Volatile solids (VS; %)<sup>1</sup></b>	19.0 ± 10.2	21.7 ± 0.1	23.6 ± 0.1	20.1 ± 0.2	26.4 ± 0.1
<b>VS/TS</b>	90.3 ± 3.5	91.4 ± 0.4	86.6 ± 0.4	92.2 ± 0.5	85.3 ± 0.7
<b>Carbohydrates (%)<sup>2</sup></b>	57.3 ± 28.8	41.4 ± 1.6	-	-	-
<b>Proteins (%)<sup>2</sup></b>	18.7 ± 6.1	15.1 ± 0.1	16.1 ± 0.3	-	-
<b>Lipids (%)<sup>2</sup></b>	12.9 ± 15.0	23.5 ± 0.3	17.5 ± 0.1	28.3 ± 1.3	-
<b>C (%)<sup>2</sup></b>	-	47.6 ± 0.5	47.2 ± 0.1	48.2 ± 0.2	46.8 ± 1.1
<b>N (%)<sup>2</sup></b>	-	3.44 ± 0.04	2.58 ± 0.05	2.50 ± 0.1	3.16 ± 0.2
<b>P (%)<sup>2</sup></b>	0.65 ± 0.17	0.54 ± 0.03	0.35 ± 0.01	-	0.52 ± 0.08
<b>Mg (ppm)<sup>1</sup></b>	2.7 ± 7.5	-	-	-	14 ± 1.0
<b>Fe (ppm)<sup>1</sup></b>	1,113 ± 1,150	-	428 ± 20	50.2 ± 3.2	766 ± 402

<b>Cu (ppm)<sup>1</sup></b>	11.2 ± 4.7	1.71 ± 0.19	-	-	31 ± 1.0
<b>Zn (ppm)<sup>1</sup></b>	38.4 ± 15.3	7.83 ± 2.61	-	-	76.0 ± 22.0
<b>Mn (ppm)<sup>1</sup></b>	27.6 ± 17.7	20.30	23.2 ± 0.4	-	60.0 ± 30.0
<b>Cr (ppm)<sup>1</sup></b>	5.2 ± 5.0	6.88 ± 0.28	-	-	3.00 ± 1.00
<b>Ni (ppm)<sup>1</sup></b>	1.2 ± 1.7	1.66 ± 0.69	-	1.46 ± 0.18	2.00 ± 1.00
<b>Co (ppm)<sup>1</sup></b>	< 9.75	< 0.06	1.30 ± 0.05	0.08 ± 0.01	-
<b>Se (ppm)<sup>1</sup></b>	-	< 0.07	-	0.13 ± 0.04	-
<b>Mo (ppm)<sup>1</sup></b>	1.3 ± 1.8	0.11 ± 0.01	2.39 ± 0.08	-	-
<b>Reference</b>	(Capson-Tojo et al., 2017b)	(Banks et al., 2012; Zhang et al., 2012a)	(VALORGAS, 2010; Yirong et al., 2015)	(W. Zhang et al., 2015a)	(Zhang et al., 2007)

1. Wet basis

2. Dry basis



As shown, FW is generally a concentrated waste stream, with total solids concentrations of 20-25% (and thus a water content of 75-80%). Among those solids, most of them (85-92%) correspond to organic matter (volatile solids), with inert contents usually below 10%. Most of the solids in FW are either carbohydrates, proteins, or lipids. The proportion of each component varies depending on the country (see Table 14.1), according to cultural habits and the predominant socioeconomic status (*e.g.* generally richer countries have higher protein intakes). Despite the differences, carbohydrates are generally the most abundant fraction (40-60%), followed by lipids and proteins (both around 15-25%). Despite their relative lower proportion, the amounts of proteins in FW are particularly relevant when considering biological processes valorisation. The main reason for this is the generation of ammonia from organic-N under anaerobic conditions, which is a crucial inhibitor of microbial growth.

As it will be discussed afterwards, this high concentration of degradable organic matter is a crucial positive feature of FW when considering the application of biological processes for its valorisation. This facilitates the potential recovery of carbon and energy, contained within carbohydrates, lipids and proteins. In addition, the high proportions of proteins in FW also make it a potential source of N and P, both crucial nutrients that need to be recovered to develop a fully circular economy (CE). This is particularly relevant for P which, unlike N, is obtained from phosphate-rich rock-mining. Around 20 million tons of P are mined every year. Most of the human-converted reactive N and mined P end-up in the environment, eroding the resilience of important Earth subsystems (*e.g.* N<sub>2</sub>O as GHG increasing radiative forcing, turbidity and anoxia in water bodies, distorting N and P cycles), and advancing rapidly towards rock-P depletion (not the case for N, but the energy-intensive Haber-Bosch process used to activate N, mostly depends on fossil fuels). Indeed, P was included in the Critical Raw Materials list of the European Union in 2017 due to the supply risk and its economic importance. P deposits are unevenly distributed and even today fertiliser quality issues frequently arise (Günther et al.,

2018). Other authors forecast that P production will peak within this century (Cordell and White, 2013) as a result of the increased need for food production, so that a future P shortage could limit food production. The relatively high concentrations of macro- and microelements that FW contains when compared to other waste streams (e.g. agricultural, dairy or food-processing industrial waste) also make the recovery of these elements attractive.

## **14.2. Food waste: an untapped resource**

### **14.2.1. Current food waste disposal processes**

Source-sorted collection is the first challenge to be faced when considering the recovery of resources from a waste stream. Nowadays, most FW is either not collected (the collection rate is around 39% in low income countries) or collected mixed with other waste streams as municipal solid waste (MSW; FW representing around half of the total MSW) (The World Bank, 2020). The lack of collection results mostly in the disposal of waste in open dumps (fate of 33% of the global waste generated). When collection is carried out, MSW is usually landfilled (37% of the total) or incinerated (11%) (The World Bank, 2020). Both practices have serious environmental consequences (e.g. leaching, greenhouse gases emission, or air pollution), allow only a limited energy recovery in the best scenario and, in any case, do not provide any nutrient recovery. Therefore, both technology developers and authorities are moving towards the limited application of these disposal approaches, developing new processes and implementing regulations to reduce their utilisation (such as the Directive 1999/31/EC on the landfill of waste). Currently, only 5.5% and 13.5% of the total MSW generated is composted or recycled, respectively (The World Bank, 2020).

Separate collection of FW is a practice implemented in few places around the globe (e.g. Seoul (South Korea), Minneapolis (USA), Oslo (Norway), Milan (Italy), Copenhagen (Denmark), or Cajicá (Colombia) (Jain et al., 2018)), but new regulations and an increasing

awareness are pushing up the number of countries following this approach. Separate collection allows the application of other practices, such as aerobic composting for production of fertilisers and organic amendments, anaerobic digestion (AD) for biogas and digestate production, and the production of animal feed (Capson-Tojo et al., 2016; Kumar et al., 2010; San Martin et al., 2016). Despite their minor current influence, these options involve some degree of resource recovery from FW, and will be further discussed in coming sections.

#### **14.2.2. Biological processes for resource recovery from FW**

Although prevention of waste production will always be the preferred strategy, a proper valorisation of FW is also paramount to achieve a sustainable food supply chain. In this context, the generation of value-added products that allow an affordable recovery of the resources contained in FW is currently being a main goal for major global players, from research institutions to food/feed producers and waste management companies. The technological feasibility of generating products such as chemicals (*e.g.* enzymes, organic acids or glycerol), materials (*e.g.* bioplastics, biopolymers or fibres), animal feeds, or fuels (*e.g.* methane, hydrogen or ethanol) has been widely proven (Capson-Tojo et al., 2018b; Kim et al., 2014; Lin et al., 2013; San Martin et al., 2016; Uçkun Kiran et al., 2014; Wang et al., 2015). Among the different options, pure chemicals generate the highest revenues ( $\approx \$1000 \cdot \text{ton biomass}^{-1}$ ), but the heterogeneous nature of FW generally imposes the need of pre and post-treatments, which results in economic outcompetition by traditional production processes (Capson-Tojo et al., 2016; Tuck et al., 2012). The direct production of feeds also leads to high revenues and is in fact commonly applied for some source-sorted FW streams (*e.g.* bakery residues). Nevertheless, this direct approach, usually simply mechanical, can only be applied to homogeneous, long-life, and pathogen free residues, excluding most FW fractions and definitely bulk FW valorisation. Transportation fuels are a relatively less profitable option ( $\$200\text{-}400 \cdot \text{ton biomass}^{-1}$ ).

<sup>1</sup>), but the enormous markets involved, the well-studied processes used, and the possibility to achieve a complete waste stabilisation foster energy recovery in the form of biofuels.

The coming sections discuss the most well-studied options for resource recovery from FW, comparing them and giving the most relevant performance numbers. Amongst the different processes that could be potentially used for bulk FW valorisation, only bioprocesses are considered, as they are generally considered to have lower costs and environmental impacts than physicochemical alternatives (Capson-Tojo et al., 2016; Girotto et al., 2015; Ma et al., 2017). In addition, due to the practical and economical limitations of waste sterilisation, processes/studies based on pure-cultures have been omitted.

#### 14.2.2.1. Aerobic composting: fertiliser production

Composting consists on the aerobic biological decomposition of organic matter, taking advantage of the heat produced biologically during the degradation process for hygienisation purposes (Walling et al., 2020). The main advantage of composting is that, through a relatively simple process (usually consists on biomass piles that are regularly aerated), solids waste streams can be properly stabilised (meaning that no further degradation occurs). Simultaneously, a nutrient-rich compost is generated as a potential value-added product that can be used as fertiliser and/or organic amendment. Nevertheless, regulatory concerns limit the applicability of this approach. As example, in the new EU regulation on fertilisers (2019/1009), composts or AD digestate are generally authorised materials, but products derived from sewage sludge, MSW, and other mixed-stream substrates are excluded. Generally, compost/digestate spreading regulations are stricter for streams originated from non-source-separated inputs (Guilayn et al., 2020). Amongst the general drawbacks of composting, it is important to highlight: (i) carbon and nutrient losses due to aerobic metabolism, (ii) no effective energy recovery, (iii) aeration needs, (iv) production of pollutant gases (*e.g.* CO<sub>2</sub>, NH<sub>3</sub> or NO<sub>x</sub>), (v)

large areas required, or (vi) odour generation.

As mentioned previously, compost is already applied as MSW treatment process (around 5.5% of the total waste (The World Bank, 2020)). Regarding FW, composting has been effectively applied for its stabilisation, achieving VS losses of 19-29% after 7-17 d (with N and C losses up to 70% and 30%, respectively) (Chang and Hsu, 2008). Nevertheless, the high water contents in FW (around 80%, see Table 14.1) usually cause mechanical issues due to its fluid consistency, and also lead to long treatment times to achieve acceptable degradation efficiencies. Because of this, co-composting of FW with drier waste streams (such as green waste) has been applied, reporting optimal VS destructions of 33% after 12 d at an initial moisture content of 60% (Kumar et al., 2010). However, mixing source-separated streams is not considered to be an optimal approach due to substrate dilution and sanitary concerns. The high protein contents in FW lead to another issue during FW composting:  $\text{NH}_3$  emissions (accounting for up to 95% of the N losses). This poses a challenge to this process, as ammonia causes odour and pollution issues (Beck-Friis et al., 2001). Gaseous emissions could potentially be avoided using additives such as phosphogypsum or superphosphate, but this approach has negative implications in terms of economic viability and process sustainability (Yang et al., 2015). Ammonia recovery techniques (*e.g.* stripping and acid absorption) would allow enhancing the economic and environmental feasibility of FW composting by generating an inorganic salt that could be crystallised to produce a dry fertiliser product.

Overall, while composting can be definitely considered as a low-cost alternative to traditional disposal methods for low-moisture waste streams (*e.g.* green or landscape waste), it is not the preferable option for resource recovery from FW, as it leads to considerable resource losses and arises different environmental concerns.

#### 14.2.2.2. Anaerobic (co-)digestion: production of biogas and digestate

AD is a well-known technology that has been applied for decades for waste stabilisation (mostly activated sludge), producing simultaneously biogas (a mixture of methane and carbon dioxide, as form of energy recovery) and digestate (residual biomass, as form of nutrient recovery). AD is a multi-stage biological process occurring under anaerobic conditions. AD is generally divided in four steps: hydrolysis (monomer production from complex molecules), acidogenesis and acetogenesis (fermentation steps in which monomers are transformed into volatile fatty acids (VFAs – *e.g.* acetic, propionic, or butyric acids –, molecular hydrogen and carbon dioxide), and methanogenesis (methane production carried out by archaea) (Appels et al., 2008). Among them, methanogenesis is generally considered as the rate limiting step, meaning that the kinetics of this latter process will be the ones limiting the overall AD kinetics. A deeper description of AD can be found elsewhere (Appels et al., 2008; Batstone et al., 2002).

The technological feasibility of FW AD has been widely demonstrated, showing that the solids contained within FW are highly biodegradable, with a biochemical methane potential (BMP) higher than other common substrates, such as activated sludge (around  $480 \text{ mL CH}_4 \cdot \text{g VS}^{-1}$  vs.  $157 \text{ mL CH}_4 \cdot \text{g VS}^{-1}$ ) (Capson-Tojo et al., 2016; Liu et al., 2012; Zhang et al., 2011). Despite the high amount of energy that can be potentially recovered via FW AD, research has shown that this process is not without challenges. These challenges arise mainly due to some of the aforementioned FW characteristics: (i) the high solid contents, (ii) the fast degradability of the contained organic matter, and (iii) the high nitrogen (protein) contents. In summary, the high solid contents favour high concentrations of potential toxic compounds (due to lack of water for dilution), the fast biodegradability might lead to process imbalance (VFA accumulation and reactor acidification), and the high protein content might lead to accumulation of ammonia-N (in the form of total ammonia nitrogen - TAN – and as free ammonia nitrogen -FAN-, see Table 14.2), a known toxic compound for methanogenic

microbes (Capson-Tojo et al., 2016).

AD is a process carried out by different clades of microorganisms and, as such, AD will fail if: (i) the specific concentrations of each toxic compound are above certain limits, or (ii) the system is operated at high organics loading rates (OLRs). The latter can cause AD failure either because a hydraulic retention shorter than the doubling time of microorganisms (leading to the so-called wash-out), or due to imbalance between VFA production rates and VFA consumption rates (causing a pH drop and the so-called reactor acidification). In the case of FW, both the lack of water and the high protein contents lead to the same issue: accumulation of  $\text{NH}_3$ , a very well-known inhibitory compound, particularly affecting the microorganisms responsible for performing methanogenesis (archaea) (Astals et al., 2018; Capson-Tojo et al., 2020b). The fact that methanogenic archaea are simultaneously the most sensitive to  $\text{NH}_3$  and the slowest-growing microbes in AD systems, might cause the process imbalance described above. If the VFAs produced during the fermentative AD steps (acidogenesis and acetogenesis) are not efficiently consumed by archaea, this will eventually lead to acid accumulation, resulting in a pH drop and reactor acidification, situation when methane production is stopped. The first four studies presented in Table 14.2 exemplify these issues. At relatively low substrate loads, all of them reported either acid accumulation (Capson-Tojo et al., 2017b), unstable operation (Zhang et al., 2012b), or an inhibited steady state (Banks et al., 2011, 2008), which is a situation in which the pH drop is avoided due to simultaneous VFAs (acids) and  $\text{NH}_3$  (base) accumulation. All those studies showed that FW AD resulted in high TAN concentrations and inefficient processes.

**Table 14.2.** Summary of some major results regarding FW (co-)AD (adapted from Capson-Tojo et al. (2016)). FW stands for food waste, OLR for organic loading rate, VS for volatile solids, TAN for total ammonia nitrogen, TEs for trace elements and GAC for granular activated carbon.

<b>FW source</b>	<b>Reactor</b>	<b>Volume (L)</b>	<b>Co-substrate</b>	<b>Additive</b>	<b>OLR (g VS·L<sup>-1</sup>·d<sup>-1</sup>)</b>	<b>Methane yield (mL·g VS<sup>-1</sup>)</b>	<b>VS destruction (%)</b>	<b>TAN (mg·L<sup>-1</sup>)</b>	<b>Reference</b>
Domestic	Pilot plant	1,500	-	-	4.1-5.7	390	67	≤ 5,200	(Banks et al., 2008)
Domestic	Industrial plant	900,000	-	-	2.5	402	-	≤ 5,000	(Banks et al., 2011)
Source sorted	Semicontinuous	35	-	-	2.0	425	84	≤ 2,500	(Zhang et al., 2012b)
Commercial	Semicontinuous	20	-	-	0.6-1.2	462-524	-	7,700-8,800	(Capson-Tojo et al., 2017b)
Source	Semicontinuous	4.0	-	TEs	2.0	470-450	-	1,500 –	(Yirong et al.,



sorted								4,200	2013)
Source Sorted	Semicontinuous + stripping	35.0	-	TEs	2.0 <sup>1</sup>	487-476	82-84	1,000- 5,100 <sup>6</sup>	(Serna-Maza et al., 2014)
University restaurant	Semicontinuous	0.3	-	TEs	2.0-6.0	504-372	69-78	≤ 2,000	(W. Zhang et al., 2015b)
Domestic	Semicontinuous	4.0	-	TEs	2.0-5.0	700-750 <sup>3</sup>	-	5,000- 6,000	(Banks et al., 2012)
Commercial	Batch	0.4	-	TEs + GAC	1.9	452	-	10,000	(Capson-Tojo et al., 2018b)
Commercial	Semicontinuous	12	-	Biochar + FeCl <sub>3</sub>	1.4-2.8	200-450	-	8,220- 8,432	(Capson-Tojo et al., 2018a)
University restaurant	Semicontinuous	2.0	Activated Sludge	-	1.0-8.0	400-0	75-0	-	(Gou et al., 2014)
Restaurant	Semicontinuous	1.8	Cattle manure	-	0.7-3.0	630-470	83-67	3,090- 3,420	(Agyeman and Tao, 2014)

Restaurant	Semicontinuous	260-280	Landscape waste	-	2.0-15 <sup>4</sup>	229-272	-	≤ 7,010	(Drennan and DiStefano, 2014)
University restaurant	Semicontinuous	80	Rice husks	-	5.0, 6.0, 9.0	446, 399, 215	82, 73, 35	-	(Jabeen et al., 2015)
University restaurant	Semicontinuous	0.3	Incineration plant leachate	-	4.0-8.3	376-506	67-82	-	(W. Zhang et al., 2015c)
Domestic	Semicontinuous		Cattle slurry, cardboard	-	2.0-4.0	50-400	-	800-5,000	(Zhang et al., 2012a)

1.  $\text{g VS} \cdot \text{kg}^{-1} \cdot \text{d}^{-1}$

2. Concentrations per kg

3. Biogas yields

4.  $\text{g COD} \cdot \text{L}^{-1} \cdot \text{d}^{-1}$

Since then, research has been directed to find solutions to solve these issues, and to improve the process performance. The most studied option has been the addition of trace elements (TEs; see Table 14.2). Several studies have shown that TEs addition (*e.g.* Ni, Se, Mo, Co and/or Mn) during FW AD can enhance the process kinetics, reducing acid accumulation and allowing higher substrate OLR (Banks et al., 2012; Serna-Maza et al., 2014; Yirong et al., 2013; W. Zhang et al., 2015b). This is explained by the predominance of hydrogenotrophic methanogenesis (a process known to be TEs demanding for enzyme synthesis) as main methane-producing pathway at high TAN levels (Capson-Tojo et al., 2020b, 2018c; De Vrieze et al., 2012; Tian et al., 2018). TEs dosage avoids the lack of these elements, allowing a more efficient/faster methanogenesis and limiting VFA accumulation.

Another type of additive that has been widely studied to enhance FW AD are carbon-based conductive materials, such as biochar or granular activated carbon (GAC). GAC has effectively used to favour VFA consumption and to enhance methanogenesis kinetics (Capson-Tojo et al., 2018b, 2017b; Dang et al., 2016). Biochar has also appeared as a cheaper alternative to GAC able to provide similar effects, but at higher dosing concentrations (Capson-Tojo et al., 2018a; Cruz Viggi et al., 2017). Carbon-based conductive materials might assist in the VFA consumption process by different mechanisms: (i) acting as surface where direct interspecies electron transfer can occur, thus favouring acid consumption via elimination of intermediate products (*e.g.* molecular hydrogen), (ii) via adsorption of inhibitors onto their surface, or (iii) by allowing the formation of biofilms (Barua and Dhar, 2017; Fagbohunbe et al., 2017). The precise mechanisms are still being researched and discussed.

A more economic approach than dosing the aforementioned additives to enhance AD performances is to mix FW with other waste streams, in the so-called co-digestion approach (opposed to mono-AD). Other than combining substrates to substitute additives (*e.g.* add TEs-rich waste streams to FW prior AD (De Vrieze et al., 2013; W. Zhang et al., 2015c)), co-

digestion leads to other positive effects, such as dilution of inhibitory compounds, macro-nutrient balance (*e.g.* N and P), or adjustment of moisture contents (Mata-Alvarez et al., 2014). In the case of FW, co-digestion has been mostly applied to dilute the nitrogen levels (thus reducing the TAN concentrations in the reactor), to supplement TEs, or to increasing the buffering capacity of the digesters (Capson-Tojo et al., 2016; Xu et al., 2017). Table 14.2 shows some representative examples of co-substrates that have been effectively used to enhance the AD performance (in terms of stability and methane productivities) when compared to mono-AD systems.

Sludge (either sewage or activated) has been commonly applied for dilution of inhibitors, and to adjust the carbon/nitrogen ratios, both strategies aiming at lowering the TAN concentrations in the digesters (Gou et al., 2014; H.-W. Kim et al., 2011; Montecchio et al., 2019; J. Zhang et al., 2015). In addition, TEs-rich sludge has also been used to stabilise FW AD (De Vrieze et al., 2013). Nevertheless, as mentioned before, the combination of source-separate waste streams is forbidden in some countries, which might limit the application of this approach (*e.g.* addition of sludge to FW as AD co-substrate is specifically forbidden in France).

Animal manure is another substrate that has been commonly used to provide buffering capacity and to reduce the concentration of easily degradable organic matter, thus reducing the extent of VFA accumulation and the pH drop (Agyeman and Tao, 2014; Dennehy et al., 2018; El-Mashad et al., 2008). It must be commented that manure addition usually resulted in systems similar to the inhibited steady-state described above, with high TAN and VFA levels and unstable operation.

Co-substrates rich in lignocellulosic compounds have been the most widely applied option, aiming at diluting the nitrogen contents and at increasing the buffering capacity in the reactors. Some examples of lignocellulosic-rich waste streams that have been efficiently applied to stabilise FW AD and/or to increase the applicable organic loads are: landscape waste

(Drennan and DiStefano, 2014), green waste (Chen et al., 2014; Karthikeyan et al., 2012), paper/cardboard waste (Capson-Tojo et al., 2017a, 2017c), or agricultural wastes (*e.g.* wheat straw (Shi et al., 2018) or rice husks (Jabeen et al., 2015)).

The anaerobic co-treatment of urban wastewater (UWW) and FW using anaerobic membrane bioreactors (AnMBRs) is a different co-digestion approach that has been recently proposed (Moñino et al., 2017; Pretel et al., 2016; Robles et al., 2018). While this option might currently face regulatory barrier in some countries, it has a great potential in regions implementing water-resource recovery facilities (WRRFs), and where wastewater and FW are already collected together (such as regions of North America using sinks with integrated food millers). Research has shown the significant positive impact of this approach in terms of methane yields (increased up to 167% by FW addition), which might improve considerably the energy self-sufficiency of future WRRFs. Indeed, this approach was identified likely to be a net energy producer, resulting in considerable cost savings.

A final stabilisation process that has been applied for FW AD is the so-called 2-stage AD. In those systems, the hydrolysis and fermentation steps (acidogenesis and acetogenesis) occur in a 1<sup>st</sup> reactor, physically separated from a 2<sup>nd</sup> stage where methanogenesis occurs (Kondusamy and Kalamdhad, 2014). In the case of FW, this has been claimed to be advantageous for two main reasons: (i) different optimal conditions can be provided to different microbial clades, and (ii) hydrogenotrophic methanogenesis (predominant at high TAN contents) can be favoured in the 2<sup>nd</sup> stage by recycling to this stage part of the biohydrogen and the carbon dioxide produced in the 1<sup>st</sup> stage (Grimberg et al., 2015; Luo and Angelidaki, 2013). Because of these advantages, 2-stage AD systems have been efficiently applied with FW as substrate, obtaining high methane yields ( $> 500 \text{ mL CH}_4 \cdot \text{g VS}^{-1}$ ) at moderate loads ( $> 2 \text{ g VS} \cdot \text{L}^{-1} \cdot \text{d}^{-1}$ ) (Ratanatamskul et al., 2015; Ventura et al., 2014; Yabu et al., 2011; Yan et al., 2016). Despite these accomplishments, studies have also shown that, when using some of the additives

described above (*i.e.* TEs), the performance of single stage and 2-stage systems is similar (Wu et al., 2015). In addition, recent studies have concluded that 2-stage processes are more sensitive to TAN/NH<sub>3</sub> toxicity, suggesting also that single stage AD might lead to more stable processes (Ariunbaatar et al., 2015; Lim et al., 2013; Rajagopal et al., 2013). These considerations, together with the higher capital and operational costs of 2-stage reactors, have limited their application, being single stage processes the most widely applied by far (Rapport et al., 2008). Nevertheless, the situation might change in the future, and 2-stage AD reactors might find niche applications with particular requirements that favour their utilisation (*e.g.* if the price of molecular hydrogen increases and its production is favoured).

Overall, AD is an effective, off-the-shelf, technology that could be directly applied for FW valorisation. In fact, it is currently being already implemented in the EU for MSW treatment, and policies exist to promote its application in many areas of the world. The environmental benefits of FW AD have been outlined by different studies applying Life Cycle Analysis (LCA), concluding that AD is preferable over other technologies for FW disposal, such as controlled landfilling, incineration or composting (Khoo et al., 2010; Xu et al., 2015). Nevertheless, if FW AD is to be globally implemented, the economic benefits must be clear and must not be dependent on local subsidies. To achieve that, cheap alternatives to the additives described above must be developed, as their overall capacity to maintain high methane yields at increasing substrate loads has been proven. Additives might be crucial for a profitable implementation of FW AD without substantial dilution with water (relatively cheap TEs commercial solutions already exist, but industrial biochar applications in AD are still to be developed) (Capson-Tojo et al., 2016). In addition, to foster the economic feasibility of the process it is imperative to develop policies that allow the commercialisation of digestate as nutrient-rich fertiliser (Guilayn et al., 2020). Regarding co-digestion, it might be a preferable alternative when effective source-sorting methods cannot be applied. Otherwise, regulatory

concerns might arise (and the value of the produced digestate might be reduced due to contamination).

#### 14.2.2.3. Anaerobic fermentation: production of hydrogen and organic acids

If the AD processes described above is stopped before methanogenesis (*e.g.* via retention time control or suppression of archaea), hydrogen and VFAs are the end products. This process, called dark fermentation (DF) has been widely used for hydrogen generation from different waste streams (Ghimire et al., 2015).

The fast biodegradability and the high carbohydrate contents of FW (see Table 14.1) make it an ideal substrate for DF. The rapid degradation causes a fast initial pH drop, which inhibits by itself methane production at high substrate loads. Regarding the carbohydrate proportions, research has shown that the hydrogen yields obtained by DF are correlated to the carbohydrate contents in the substrates, as monosaccharides (products of carbohydrate hydrolysis) are the biologically preferred fermentation substrates (Capson-Tojo et al., 2016; Guo et al., 2014; Kobayashi et al., 2012). Because of the suitability of FW for hydrogen production, several studies have been carried out on the topic. As it can be observed in Table 14.3, the highest hydrogen yields presented are within the top values common for DF processes, with numbers up to  $0.19 \text{ g COD} \cdot \text{g VS}^{-1}$  (or around  $250 \text{ mL H}_2 \cdot \text{g VS}^{-1}$ ) (Moscoviz et al., 2018). This confirms the potential of FW as substrate for hydrogen production via DF.

**Table 14.3.** Summary of some major results regarding FW fermentation (adapted from Capson-Tojo et al. (2016)). FW stands for food waste, OLR for organic loading rate, VS for volatile solids, TAN for total ammonia nitrogen and VFA for volatile fatty acid.

FW source	Product aimed	Reactor	Volume (L)	T (°C)	OLR (g VS·L <sup>-1</sup> ·d <sup>-1</sup> )	Product yield (g COD·g VS <sup>-1</sup> ) <sup>1</sup>	VS destruction (%)	TAN (mg·L <sup>-1</sup> )	VFA (mg COD·L <sup>-1</sup> )	Inoculum pretreat.	pH	Reference
Cafeteria	H <sub>2</sub>	Batch	3.0	35	-	0-0.11	≤ 50	-	≤ 33,991	FW heated	5.0	(Kim et al., 2009)
University restaurant	H <sub>2</sub>	Sequential batch	4.5	35	-	0.01-0.06	55-76 <sup>2</sup>	-	19,700-25,800	Heat	> 5.3	(Kim et al., 2008)
Synthetic	H <sub>2</sub>	Semicontinuous	3.0	34	11-45	0.02-0.01	53-47	78-15	8,500-6,100	No	5.5	(Redondas et al., 2012)
Cafeteria	H <sub>2</sub>	Semicontinuous	5.0	55	70-125 <sup>3</sup>	0.05-0.08	-	-	15,399-20,933	Heat	5.5	(Lee et al., 2014)
Kitchen waste	H <sub>2</sub>	Semicontinuous	3.0	35	27-100 <sup>3</sup>	0.05-0.07 <sup>2</sup>	56-23 <sup>2</sup>	320-	10,900-	-	5.5	(Li et al.,



								670	8,900			2011)
University restaurant	H <sub>2</sub>	Semicontinuous	0.7	35	-	0.06-0.19	-	-	493-1,084	Heat	5.0	(Reungsang et al., 2013)
University restaurant	H <sub>2</sub>	Semicontinuous	0.7	37	-	0.05-0.12	33-53	nr	16,000-25,000	-	6.0	(Jang et al., 2015)
Synthetic	Mixed VFA	Batch	4.5	35	-	0.47-0.80	42.5-50.0	< 1,800	26,100-62,640	-	6.0	(Wang et al., 2015)
University restaurant	Mixed VFA	Batch	5.0	5-65	-	0.05-0.67	-	-	-	-	4.0-12.0	(Y. Chen et al., 2013)
University restaurant	Mixed VFA	Batch	0.5	30	-	0.67-0.76 <sup>4</sup>	-	-	22,000-23,900	-	6.0	(Shen et al., 2016)
Cafeteria	Mixed VFA	Semicontinuous	2.0	25-45	5, 9, 13 <sup>5</sup>	0.22-1.50	-	0-51	5,000-30,000	-	5.0-6.0	(Lim et al., 2008)
Fruit/vegetable waste	Lactic acid	Semicontinuous	4.5	35	11	-	-	68	10,000-29,500	-	4.0, 5.0	(Y. Wu et al., 2015)

1. Estimated assuming normal conditions for gases (COD equivalent of 1,400 mL·g COD<sub>H2</sub><sup>-1</sup>)

2. VSS (% or base unit)
3.  $\text{g COD} \cdot \text{L}^{-1} \cdot \text{d}^{-1}$
4.  $\text{g} \cdot \text{g VS}_{\text{removed}}^{-1}$
5.  $\text{g TS} \cdot \text{L}^{-1} \cdot \text{d}^{-1}$

As aforementioned, for an effective fermentation, it is required to avoid the presence of archaea to minimise hydrogen/VFA consumption. This is mainly achieved by two means: (i) working in continuous systems at retention times not allowing archaeal growth (generally below 2 d), and (ii) pretreating the microbial inoculum, mostly via thermal processes (Elsamadony et al., 2015; Ghimire et al., 2015). In many (semi)-continuous experiments both approaches are combined, using initially pretreated inocula and feeding at OLRs over  $8 \text{ g VS}\cdot\text{L}^{-1}\cdot\text{d}^{-1}$  (thus working at low retention times and leading to archaea wash-out).

A main challenge of FW DF is that the pH needs to be kept within the desired pH values (6.5-7.5) for hydrogen production, as it regulates both the product profile and the obtained hydrogen yields (Ismail et al., 2009; Nazlina et al., 2009). As VFAs are produced during fermentation, the pH naturally tends to go down, reason why buffer and basic solutions are commonly dosed in reactors (Ghimire et al., 2015; Han et al., 2015; Jang et al., 2015).

As an alternative to external chemical dosing, co-fermentation has been used for increasing the buffering capacity during FW DF. Low biodegradable, highly-buffered, co-substrates have been effectively used to avoid excessive pH drops. Examples are cardboard (Capson-Tojo et al., 2018d), paper waste (Valdez-Vazquez and Poggi-Varaldo, 2009), olive husks (Pagliaccia et al., 2016), or white mud (Zhang and Wang, 2013). Nevertheless, the generally recalcitrant nature of these compounds usually reduces the applicable OLRs, increasing the reactor volumes and reducing the treatment capacities.

In addition to hydrogen, fermentation can also towards maximizing the production of organic acids, simply by working at lower pH values (5.5-6.5 for mixed VFA production or around 4 if lactic acid is the desired product) (Dahiya et al., 2015; Wainaina et al., 2019; Y. Wu et al., 2015; Zhang et al., 2020). High metabolite yields have been achieved (up to  $0.8 \text{ g COD}\cdot\text{g VS}_{\text{fed}}^{-1}$ ), proving the technological feasibility of this approach. As for DF, in this strategy pH control is generally also required, reason why co-fermentation has also used (although to an

even lower extent) (H. Chen et al., 2013; Y. Chen et al., 2013).

The production of lactic acid from FW has recently appeared as a particularly interesting option, mainly due to the high yields and the higher purities achieved in the product when compared to mixed VFAs (Gu et al., 2018; Y. Wu et al., 2015). In addition, it has been found that native bacteria in FW can spontaneously start lactic fermentation efficiently, without the need of inoculum (D.-H. Kim et al., 2011). Finally, the required lower pH during lactic acid fermentation demands a much lower buffer dosage than mixed VFA processes. Co-fermentation of FW and activated sludge has also been tested for lactic acid production (Li et al., 2018), concluding that FW mono-fermentation is a more suitable process.

Although the effective industrial implementation of DF from FW is anecdotal due to the low production rates and the current low hydrogen prices, the extensive scientific efforts performed make this technology worth to mention (Moscoviz et al., 2018). In addition, if the application of hydrogen as future fuel and energy carrier is extended, its price might increase considerably, favouring the economic feasibility of DF processes. Regarding acid production, a similar situation is faced, where current fossil-fuel derived chemicals are more price competitive than biosourced compounds. In addition, as the final product purification is still a challenge, a major effort must be made to increase both the concentration and the selectivity of the acid desired in the mixed products. Future market/technology developments and the generation of products with higher value (*e.g.* longer chain fatty acids such as caproic acid via chain elongation) might favour acid production as valorisation route.

#### 14.2.2.4. Novel approaches: production of single-cell protein and pure chemicals

New technologies are driving a shift in the environmental sector, moving away from simple waste disposal towards the implementation of resource recovery alternatives. As the name implies, these processes must allow the recovery of materials and energy from waste streams.

However, if these novel technologies are ever to be implemented, they need to be economically competitive and for that, they need to generate value-added products that balance out their higher costs when compared to traditional alternatives (Batstone et al., 2015). To achieve this, research is being performed to develop alternatives to the production of cheap energy carriers (such as biomethane via AD), aiming at generating more valuable products. Among the latter, single-cell protein (SCP) and biosourced chemicals/materials are gaining momentum.

SCP produced from waste can be used as animal feed, or even for direct human consumption (Jones et al., 2020). It has been estimated that recycling of the nutrients in waste into SCP could represent globally around 8% of the current nitrogen losses, and could provide 25% of the annual phosphorous fertiliser production (Matassa et al., 2020). Examples of SCP that have already been commercialised for human consumption are Quorn®, Vegemite®, or dry yeast flakes from brewery processes (several products available).

SCP can be directly produced by growing microbial biomass on waste streams. Different approaches are being developed, such as the growth of aerobic heterotrophic bacteria in food and beverage effluents (Muys et al., 2020), or the growth of purple phototrophic bacteria in MSW hydrolysates and wastewaters (Allegue et al., 2020; Capson-Tojo et al., 2020a). Other than bacteria, edible fungal biomass from VFAs derived from FW has also been produced (Wainaina et al., 2020). Very few studies have been tested so far using FW directly as substrate. Despite the great potential of this approach, challenges such as the need of generating a safe product, the lack of public acceptance, or missing/outdated recycling legislations, arise. If these technologies are to succeed for feeding purposes, the final products must be pathogen/contaminants free and offer an appealing nutritional value. Regarding human consumption, whether people will accept or not an accelerated version of the biochemical processes than nature uses to deal with waste, is a question that will be answered in the coming years. We should always keep in mind that “waste” is merely a human definition, irrelevant in

the natural environment.

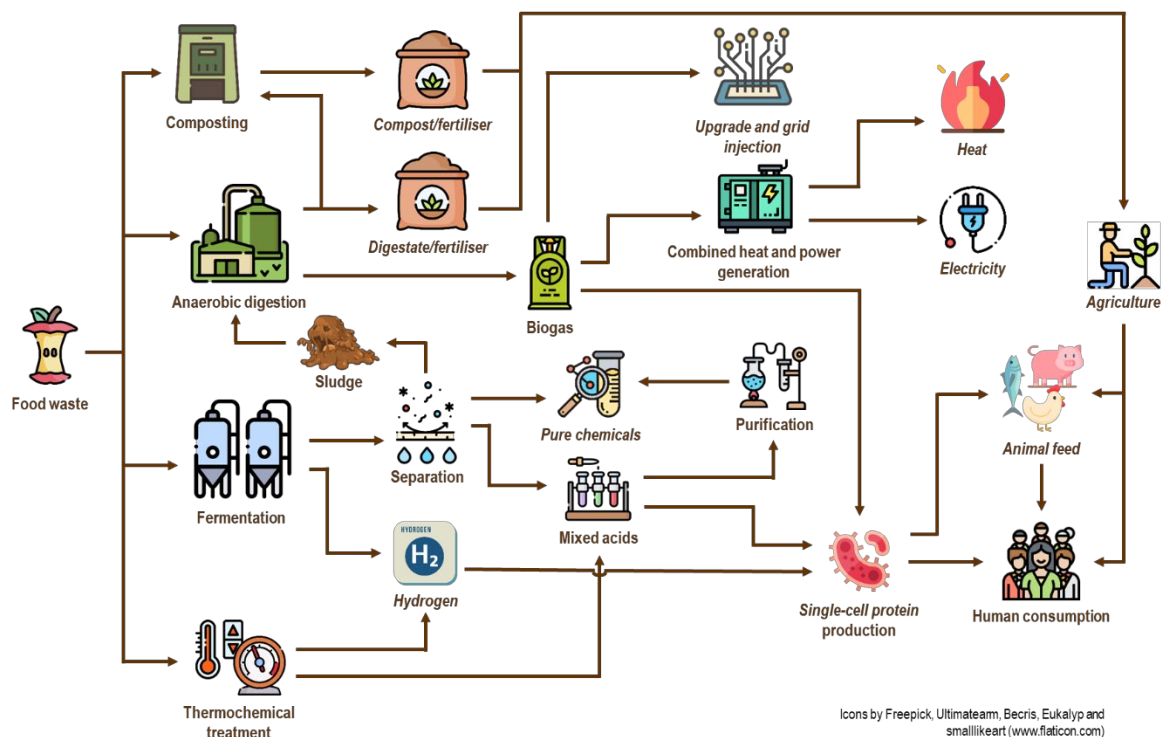
An alternative to direct resource recovery from wastes consist on the previous production of clean gaseous substrates (*e.g.* H<sub>2</sub>, CH<sub>4</sub>, CO<sub>2</sub>, CO, NH<sub>3</sub> or P<sub>2</sub>) via biological (*e.g.* AD or DF) or physicochemical processes (*e.g.* thermochemical gasification), which can afterwards be used as substrate for production of SCP (De Vrieze et al., 2020; Matassa et al., 2020). Although more complex (and thus expensive) than direct waste conversion, this approach has the advantage of avoiding any safety concern related to the presence of pollutants/pathogens in the waste. Although this might seem trivial, ensuring the generation of a clean product is the main challenge of SCP from wastes. Not only the safety and the applicability of the products depend of this, but also the regulatory benchmark to be developed and the social acceptance of the generated SCP. All of those are crucial criteria that must be fulfilled if SCP is to be a relevant source of recovered resources worldwide. SCP production form waste-derived energy-rich gases represents an option to produce safe protein-rich microbial biomass with a great future ahead.

Other than SCP, the production of pure, high value-added, biochemicals from FW is an option being currently researched. In this case, the challenge for most compounds is not in the production step itself (see Section 2.2.3), but in the separation and purification steps. Other than traditional separation methods such as distillation, precipitation, adsorption, or extraction (Aghapour Aktij et al., 2020), novel, cleaner, cost-effective alternatives are also being tested, such as the application of green solvents such as supercritical CO<sub>2</sub> (Campalani et al., 2020) or selective ionic liquids (Escudero et al., 2020), or the utilisation of membranes (Aghapour Aktij et al., 2020). These novel alternatives have the potential of reducing separation costs to numbers where the high value of the products might make the overall process profitable.

### **14.2.3. The future of biological FW valorisation: the environmental biorefinery?**

With increasing concerns about environment degradation, food and energy security, and climate change, the focus on FW treatment is shifting towards developing more energy-efficient and cost-effective systems. The integration of different technologies (such as the presented above) within a comprehensive, optimised, resource recovery facility has a great potential to favour a sustainable FW valorisation and management. The so-called WRRFs and biorefineries are based on this principle, and are expected to play a main role in Circular Economy (Moscoviz et al., 2018; Solon et al., 2019; Venkata et al., 2016). Although nowadays the economic feasibility of this approach is still questionable, future technological and societal developments will surely favour its implementation.

Different examples of potential integrations are presented in Figure 14.2. An example widely discussed in the literature is the so-called acidogenic biorefinery (Capson-Tojo et al., 2016; Kim and Kim, 2013; Kim et al., 2016; Venkata et al., 2016). In this approach, FW is initially used to produce value-added organic acids (*e.g.* via fermentation or thermochemical pre-treatment), which are afterwards recovered. The remaining sludge (still to be stabilised) is afterwards treated via AD, further recovering energy and carbon in the form of biogas. Finally, nutrients could be recovered by direct digestate application as fertiliser.



**Fig. 14.2.** Scheme representing different potential bio-based routes for generation of value-added products from FW. For simplification, the applications of hydrogen as product are omitted.

Another step that could be integrated within the acidogenic biorefinery is the production of SCP from the mixed organic acids produced during fermentation, allowing nutrient recovery and avoiding purification steps (Alloul et al., 2018; Capson-Tojo et al., 2020a; Puyol et al., 2017; Wainaina et al., 2020). In addition, the biogas produced during AD or fermentation could be also used as gaseous substrate to produce SCP, increasing the value of the final product and further favouring nutrient recovery (Bernaerts et al., 2020; Matassa et al., 2020). Photosynthetic organisms (*e.g.* purple phototrophic bacteria and microalgae) are promising alternatives for nutrient recovery via SCP production, not only due to their ability to assimilate nutrients (and carbon in the case of purple bacteria) and their smaller carbon footprint compared to other technologies, but also because of the competitive biomass yields (Capson-Tojo et al., 2020a;



Chen et al., 2019; Garrido-Cardenas et al., 2018; Guldhe et al., 2017; Lu et al., 2019; Romero-Villegas et al., 2018; Santos and Pires, 2018). Nevertheless, the economic performance of these processes still needs to be improved.

Other than fertiliser or SCP production (both potentially limited due to safety and regulatory issues), different post-treatment technologies that could broaden the nutrient recovery potential of anaerobic biorefineries are being explored. There are different technologies that can be used for nutrient recovery from AD supernatants (Robles et al., 2020). Forward osmosis (FO), hollow-fibre membrane contactors (HFMC), electrodialysis (ED), or bioelectrochemical system (BES) can be efficiently applied for N and/or P recovery, and can replenish resources for biofertiliser production, thus overcoming environmental impacts due to fertiliser production. For instance, N can be recovered as ammonium sulphate by using sulphuric acid solution by HFMC, while P can be fractionated as phosphoric acid by ED (Xie et al., 2016). However, the economic competitiveness of membrane-based technologies for resource recovery must be improved to reach full-scale implementation. In this respect, membrane fouling is a big challenge, since it increases the energy demand, and thus jeopardises the economic feasibility (Yan et al., 2018). In addition, filtration processes are not applicable in high-solids streams. Another option for nutrient recovery is crystallisation, which at present is one of the most promising technologies for recovering P and N from wastewater (Li et al., 2019; Peng et al., 2018). Unlike other nutrient recovery processes (biological, electrochemical, ion exchange, or membrane systems), struvite crystallisation shows notable recovery rates with acceptable economic efficiency and generates a marketable product with a limited environmental impact (Peng et al., 2018). However, when economic reasons are weak, governments would need encourage P recovery by providing incentives to recover products by means of legislation. Other than the methods mentioned above, a number of studies have also assessed the potential nutrient recovery from final waste streams by different techniques such

as incineration, ammonia stripping and absorption, adsorption and ion exchange, etc. (Robles et al., 2020).

The recovery of FW through an environmental biorefinery approach thus requires the integration and coupling of different unitary (bio)processes. Research work is, of course, essential to improve our understanding of the key mechanisms involved in each process, in order to find levers to optimise them individually. However, it is also essential to find compromises between each optimised system. Process modelling tools, coupled with environmental, economic and social analysis approaches, must be the basis for the development of the new systems for FW valorisation and management.

### **14.3. Summary**

The need for sustainable development is highlighting the potential and relevance of FW as substrate for resource recovery (*e.g.* carbon, nutrients or energy). Factors driving the development of this approach are: (i) the enormous amounts of FW being produced, (ii) its ubiquitous nature, and (iii) its appropriate physicochemical characteristics. In this respect, the high content of degradable organic matter facilitates the potential recovery of resources from FW via biological valorisation.

Source-sorted collection is the first challenge to be faced when considering the recovery of resources from a waste stream. Separate collection enhances the application of biological practices, such as aerobic composting for production of fertilisers and organic amendments, AD for biogas and digestate production, and DF for hydrogen and VFAs production. Furthermore, SCP and bio-sourced chemicals/materials production from novel technologies is currently gaining momentum, although there is still room for improvement to generate value-added products that balance out implementation and operating costs when compared to traditional alternatives. Hence, the technological and economic feasibility of generating products such as

chemicals (*e.g.* enzymes, organic acids or glycerol), materials (*e.g.* bioplastics, biopolymers or fibres), animal feeds, or fuels (*e.g.* methane, hydrogen or ethanol) from different valorisation techniques needs further evaluation.

The integration of different technologies within a biorefinery platform also has a great potential to favor a sustainable FW valorisation and management. Acidogenic biorefineries would allow producing and recovering value-added organic acids (*e.g.* via fermentation or thermochemical pre-treatment), while the remaining sludge could be further valorised, recovering energy and carbon in the form of biogas. Nutrients could be recovered by direct digestate application as fertiliser, or integrating other technologies such as membrane-based systems (*e.g.* FO, HFMC, ED, or BES), or crystallisation, among others. SCP production also represents an attractive alternative to be integrated within acidogenic biorefineries, allowing nutrient recovery and avoiding purification steps.

Nonetheless, research is still needed to improve our understanding of the key mechanisms involved in different unitary (bio)processes, eventually aiming at developing optimised environmental biorefinery platforms. In this respect, process modelling tools, coupled with environmental, economic and social analyses, must be the basis for the development of these new systems for FW management and valorisation.

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