

## **CHAPTER 11**

# **A Logistics Analysis for Advancing Carbon and Nutrient Recovery from Organic Waste**

*Edgar Martín-Hernández,<sup>1,2</sup> Apoorva M Sampat,<sup>3</sup> Mariano Martin,<sup>2</sup>  
Victor M Zavala<sup>3</sup> and Gerardo J Ruiz-Mercado<sup>4,\*</sup>*

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### **1. Circular Economy, Waste to Energy and Chemicals, and Material Recovery**

Modern societies are characterized by the generation of large amounts of waste, arising from the production of goods and services to satisfy social demands. The traditional manufacturing pattern is one-way linear, following the raw material extraction from the environment, manufacturing, usage, disposal of goods, energy consumption, and discarding the generated residues along the linear path. Hence, when considering the definition of sustainable development formulated by the World Commission on Environment and Development (WCED) in 1987, as “a development that meets the needs of the present without compromising the ability of future generations to meet their own needs”, the linear production is an unsustainable production model, depleting the natural resources and degrading the environment.

Therefore, one of the main challenges of the current societies is to find processes and technologies that can transform the traditional linear production paradigm into a circular structure, where the materials are recovered to be reused, remanufactured, repaired, and finally closing the loop when these generated residues can be reused as raw material again. These circular production systems are aligned with the definition of a circular economy, which is formally defined by Korhonen et al. (2018) as follows:

“Circular economy is an economy constructed from societal production-consumption systems that maximizes the service produced from the linear nature-society-nature material and energy throughput flow. This is done by using cyclical materials flows, renewable energy sources, and cascade-type energy flows. Successful circular economy contributes to all the three dimensions of sustainable development. Circular economy limits the throughput flow to a level that nature tolerates and utilizes ecosystem cycles in economic cycles by respecting their natural reproduction rates”.

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<sup>1</sup> Oak Ridge Institute for Science & Education, hosted by U.S. Environmental Protection Agency, Office of Research & Development, Cincinnati, Ohio 45268, U.S.

<sup>2</sup> PSEM3 Research Group, Department of Chemical Engineering, University of Salamanca. Pza. de los Caídos 1-5, 37008 Salamanca, Spain.

<sup>3</sup> Department of Chemical and Biological Engineering, University of Wisconsin-Madison, 1415 Engineering Dr., Madison, Wisconsin 53706, U.S.

<sup>4</sup> Office of Research and Development, National Risk Management Research Laboratory, U.S. Environmental Protection Agency, 26 West Martin Luther King Drive, Cincinnati, Ohio 45268, U.S.

\* Corresponding author: ruiz-mercado.gerardo@epa.gov

Therefore, the circular economy pursues both environmental and economic objectives. The environmental objectives are to maximize the generation of valuable products, minimize the needs of raw materials extracted from nature, and reduce releases to the environment by maximizing the reuse and recycling of materials. Regarding the economic goals, these consist in the reduction of raw materials and energy costs by applying material circular cycles and mitigating the costs associated with waste and emission management.

Within the purview of circular economy, the management of carbon and nutrients from organic waste can have a huge contribution due to the large amounts of organic residues generated from agricultural activities (e.g., livestock), food waste, biosolids, etc. In addition, the concept of waste to energy and value-added chemicals, which pursues the development of processes for the recovery of energy and production of high added-value chemicals from all kind of residues, can be applied to organic waste in order to achieve a circular economy framework. To illustrate this, a diagram of a circular economy system built around organic waste is shown in Figure 1. Particularly, an anaerobic digestion process can be used to decompose the organic matter and produce biogas, constituted mainly of methane ( $\text{CH}_4$ ) and carbon dioxide ( $\text{CO}_2$ ). With proper treatment to recover the  $\text{CO}_2$ , the resulting high  $\text{CH}_4$  purity gas can be used as a renewable alternative to the fossil-based natural gas (NG) in the production of power and heat. In addition, the sub-product from the anaerobic digestion, so-called digestate, is rich in nutrients (particularly nitrogen and phosphorous) that can be recovered and used as fertilizer for crops, thus closing the nutrient cycle loop.

The livestock agricultural sector is one of the largest economic activities in the world, as it must meet the needs of a demanding society. Additionally, the increasing income-spending potential of the middle class in developing countries has increased the demand for dairy and beef products. This results in some areas where livestock activities generate more GHGs than the transportation sector,<sup>1</sup> nutrient pollution, and other environmental challenges.

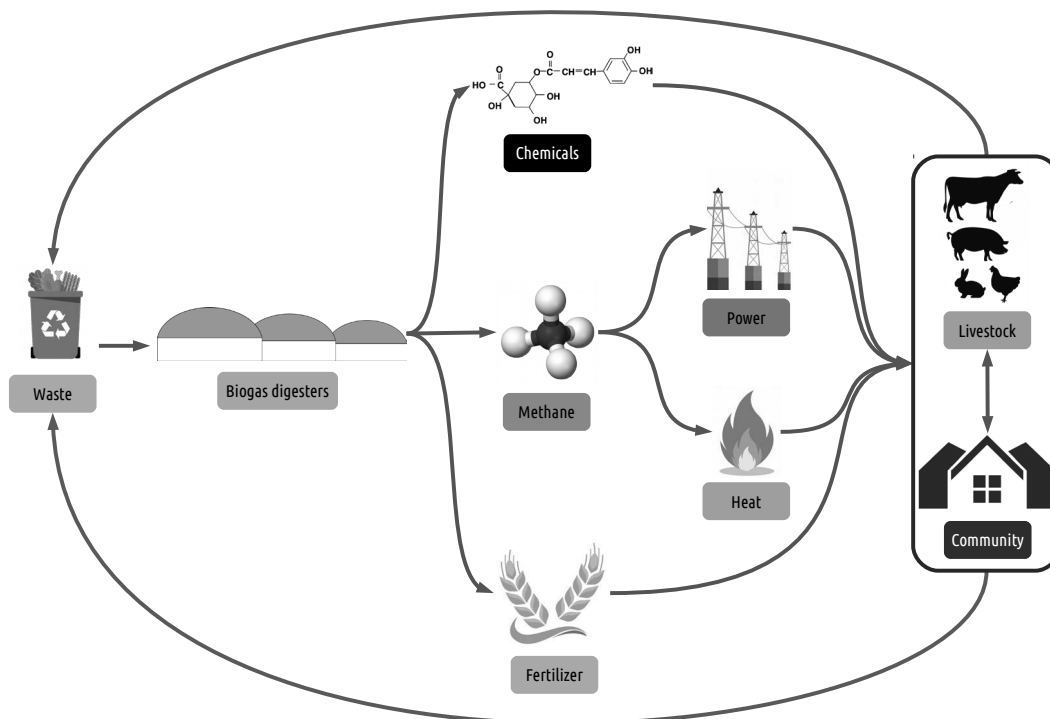
In developed societies, waste valorization to energy and material recovery represents a business opportunity for circular economy. Furthermore, organic waste management provides a great opportunity towards the production of sustainable resources and energy (WEC, 2016) and the capability of replacing fossil-based fuels. For instance, capturing carbon through the obtention of bio-based chemicals derived from the biogas generated after the anaerobic digestion of the organic wastes offers an alternative for replacing the generation of electricity and heat from fossil fuels with renewable sources throughout the production of bio-methane. It is reported that about 250 operational projects in the U.S. (Nov. 2017)<sup>2</sup> have avoided the emissions of  $3.2 \times 10^6$  tons  $\text{CO}_2\text{e}$  (equivalent to annual emissions from 680,000 cars)<sup>3</sup> and generated  $1.03 \times 10^6$  MWh of energy (enough to power 96,000 U.S. homes/yr).<sup>4</sup> Other studies have evaluated the potential of some regions to meet all their NG needs by using biogas instead (Taifouris and Martin, 2018). Therefore, waste-to-energy initiatives have gained support in the context of a circular economy philosophy to enhance the development of sustainable process alternatives (Korhonen et al., 2018). Among the organic waste treatment technologies, anaerobic digestion is deemed as an interesting and promising alternative that serves a double objective when processing residues, of mitigating the potential environmental and human health issues and producing valuable products which are incorporated into the economic cycle in the form of energy and chemicals.

<sup>1</sup> <http://www.fao.org/newsroom/en/news/2006/1000448/index.html>.

<sup>2</sup> <https://www.epa.gov/agstar/agstar-data-and-trends>.

<sup>3</sup> A typical passenger vehicle emits about 4.7 metric tons of carbon dioxide per year, <https://www.epa.gov/greenvehicles/greenhouse-gas-emissions-typical-passenger-vehicle>.

<sup>4</sup> In 2016, the average annual electricity consumption for a U.S. residential utility customer was 10,766 kWh, <https://www.eia.gov/tools/faqs/faq.php?id=97&t=3>.



**Figure 1.** Circular economy generated around organic waste.

## 2. Biogas Upgrading and Carbon Capture

### 2.1 Bio-methane as a renewable source of energy

The biogas generated through anaerobic digestion of waste is mainly composed of  $\text{CH}_4$ ,  $\text{CO}_2$ , small amounts of impurities (ammonia and hydrogen sulfide), and moisture. After removing these impurities, biogas represents a renewable source of  $\text{CH}_4$  and  $\text{CO}_2$ . Biogas can be employed either to produce energy, as it is currently implemented in many industrial facilities, or as raw material to produce chemicals through a syngas/dry reforming path.

As a source of energy, the usage of biogas can be classified according to its quality. Raw biogas can be used for the co-generation of heat and electricity in combined heat and power (CHP) units for onsite consumption, mainly to supply heat to the digester unit and produce hot water. In addition, it is possible to use biogas in gas turbines and power generators if upgraded by reducing the concentration of  $\text{CO}_2$  and obtaining high purity methane (Somehsaraei et al., 2014; Reddy et al., 2016; León and Martín, 2016). Additionally, the large infrastructure available for the transportation and distribution of NG in Europe (Entsog, 2015) or the USA (EIA, 2018), can support the use of biomethane as a renewable alternative to NG. Likewise, high-quality biomethane can be compressed or liquefied to be transported using trucks or railways if a pipeline grid is not available. Finally, a remarkable feature of compressed or liquefied biomethane is its use as transportation fuel, replacing compressed-NG (CNG) and liquefied-NG (LNG).

There are several pathways for upgrading biogas. For example, it is possible to hydrogenate the  $\text{CO}_2$  content of the biogas into  $\text{CH}_4$ . This process is widely used as a purification step for ammonia production and has been evaluated for the production of biomethane from biogas at experimental level (Stangeland et al., 2017; Schidhauer and Biollaz, 2015) and lately full process engineering, including techno-economic analysis (Curto and Martín, 2019), have been reported. The main issue associated with the hydrogenation of  $\text{CO}_2$  to obtain renewable biomethane is the high need for hydrogen from renewable energy sources

instead of using fossil fuel sources since its production is energy intensive (Davis and Martin, 2014a, b). The high investment costs for renewable energy production using either photovoltaic panels and/or wind turbines limit the placement of these facilities to favorable locations where solar fields and wind farms can achieve competitive production costs of  $\text{CH}_4$  from  $\text{CO}_2$  (de la Cruz and Martín, 2016). Additionally, the recovered  $\text{CO}_2$  has enough purity to be used as a raw material for other chemical processes.

Regarding the use of biogas as raw material to produce chemicals, some recent studies evaluate different routes to produce bio-synthesis gas (Hernández et al., 2017), methanol for biodiesel, and as a feedstock for algae-based oil production (Hernández and Martín, 2017). However, the investment and production costs of these chemicals through these novel routes are high, resulting in using biogas as a primary energy source alone.

## 2.2 Technical alternatives for biogas-carbon management

The common  $\text{CO}_2$  capture techniques are based on separation processes, such as gas-liquid chemical absorption, gas-solid physical adsorption, and permeation processes using membranes. Currently, the most mature technologies for  $\text{CO}_2$  capture, based on the studies available in the literature, are amine scrubbing, pressure swing adsorption (PSA), and membrane separation systems. Water scrubbing is another technology that is widely used for  $\text{CO}_2$  capture, but it has some major disadvantages. Particularly, the air used for the degassing of water (Budzianowski et al., 2017) exits the unit with a high concentration of  $\text{CO}_2$  that should be treated before being released to the atmosphere. Furthermore, this  $\text{CO}_2$  can be used after being recaptured through additional processing steps.

### 2.2.1 Chemical absorption: Amines

For the  $\text{CO}_2$  absorption using amines solution, the gas stream is passed counter-currently to an aqueous alkali solution containing the amine in order to promote a chemical absorption of the  $\text{CO}_2$ . As described in Figure 2, this process is carried out in a two-exit absorption column. The gas phase output stream is composed of the purified gas stream, while the liquid phase output stream consists of the aqueous solution with the absorbed  $\text{CO}_2$ . The aqueous stream is sent to a regeneration column in order to perform the desorption process by heating the solution. Even though the absorption process does not require external energy at ambient temperature, the regeneration process is energy intensive (Ünveren et al., 2017).

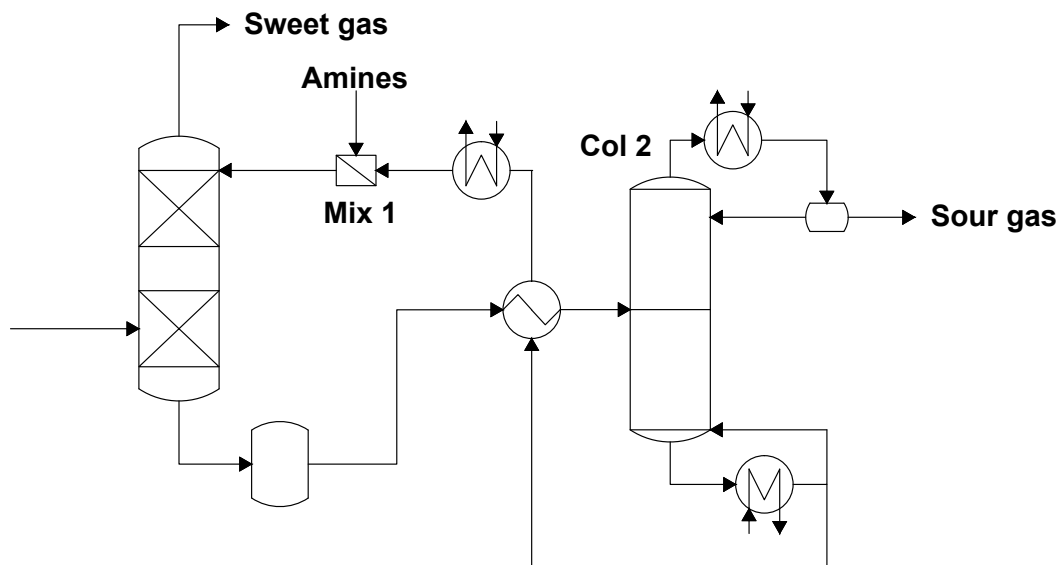


Figure 2. Flowsheet describing the amine absorption process.

The CO<sub>2</sub> absorption systems using amines typically operate at a temperature range of 25–30 °C, partial pressures above 0.05 bar, and removal yields of 90%–95% (Zhang and Chen, 2013). However, the high concentration of CO<sub>2</sub> in the biogas, compared to post-combustion gases, results in lower operating pressures, reducing the compression costs.

Different amines, like monoethanolamine (MEA), diethanolamine (DEA), and methyl diethanolamine (MDEA) are among the common amines used for CO<sub>2</sub> capture, because of their high CO<sub>2</sub> affinity. The selection of the optimal amine to minimize the overall operating costs should be made considering several important aspects, such as the CO<sub>2</sub> to amine molar ratio, the heat of reaction (associated to the reactor refrigeration cost), and the absorption solution concentration and its price (Nuchitprasittichai and Cremaschi, 2013).

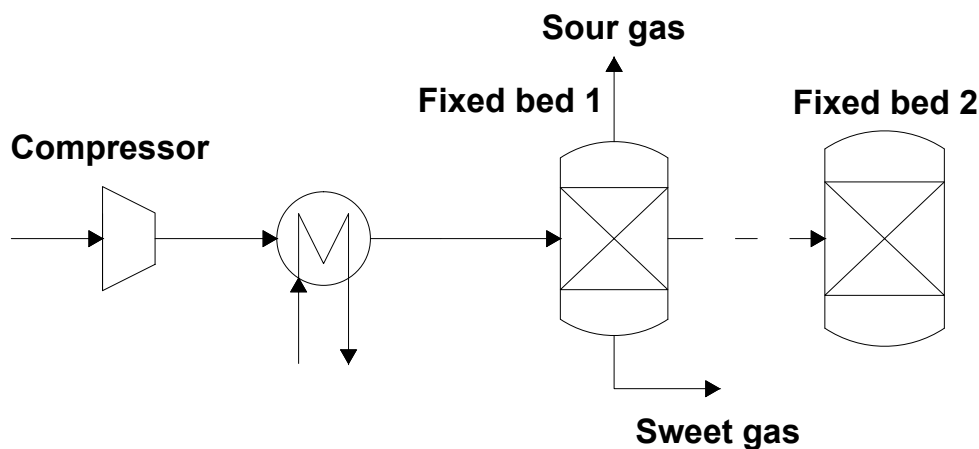
Amines absorption is a relatively simple CO<sub>2</sub> capture technology, which does not require the development of complex or novel equipment. The requirement of energy involved in the regeneration column is the largest share of the operating cost.

### 2.2.2 Pressure swing adsorption

A pressure swing adsorption (PSA) operation is based on physical adsorption at moderate pressure and low-pressure regeneration at a constant temperature. As shown in Figure 3, in order to perform the CO<sub>2</sub> adsorption, the raw gas is compressed prior to the adsorption stage. Then, the compressed gas is introduced into a column filled with the solid adsorbent material, forming a fixed bed. The gas is flowing across the bed, interacting with the adsorbent, and carrying out the CO<sub>2</sub> adsorption on the fixed bed through reaching a certain partial pressure for CO<sub>2</sub>. After the bed reaches the saturation point, the operation stops, and the unit undergoes a desorption step. The desorption step consists of a concurrent flow of a gas stream with low CO<sub>2</sub> partial pressure which drags the CO<sub>2</sub>. Commonly, PSA systems are composed of twice the amount of design-required adsorption columns, so while one-half of the units are in use, the other half are being regenerated.

Some of the most common adsorbent materials include zeolites, such as zeolites 13X and 4A. The CO<sub>2</sub> removal before the breakpoint using zeolites 13X and 4A is almost total, leaving only traces of CO<sub>2</sub> in the exit gas (Hauchhum and Mahanta, 2014). Additionally, some experimental results show that CO<sub>2</sub> adsorption onto zeolites can be modeled through theoretical isotherms, like the Langmuir isotherm (Hauchhum and Mahanta, 2014).

The adsorption capacity of the zeolites is directly related to the CO<sub>2</sub> partial pressure. Therefore, a system of compressors with an intermediate cooling system should be implemented. However, more detailed studies should be carried out in order to determine the optimal trade-off between compression cost and carbon capture efficiency.



**Figure 3.** Flowsheet describing the pressure swing adsorption process.

### 2.2.3 Membrane separation systems

Membrane separation technologies are based on the permeability of different gases through a membrane. The species with higher permeability can cross the membrane, forming an outlet stream, i.e., permeated, while the species with lower permeability cannot go through the membrane barrier, being evacuated from the unit through an outlet stream, i.e., retentate.

As the driving force of the process is the gradient in the species concentration on both sides of the membrane, certain partial pressure values for the permeated compounds should be achieved before introducing the gas stream in the membrane unit. As shown in Figure 4, membrane systems consist of compressor units followed by a set of membrane modules. For the design of membrane systems, two variables should be considered: The membrane material and the arrangement of the units, since the membrane units can be arranged in different configurations such as single-stage or multi-stage

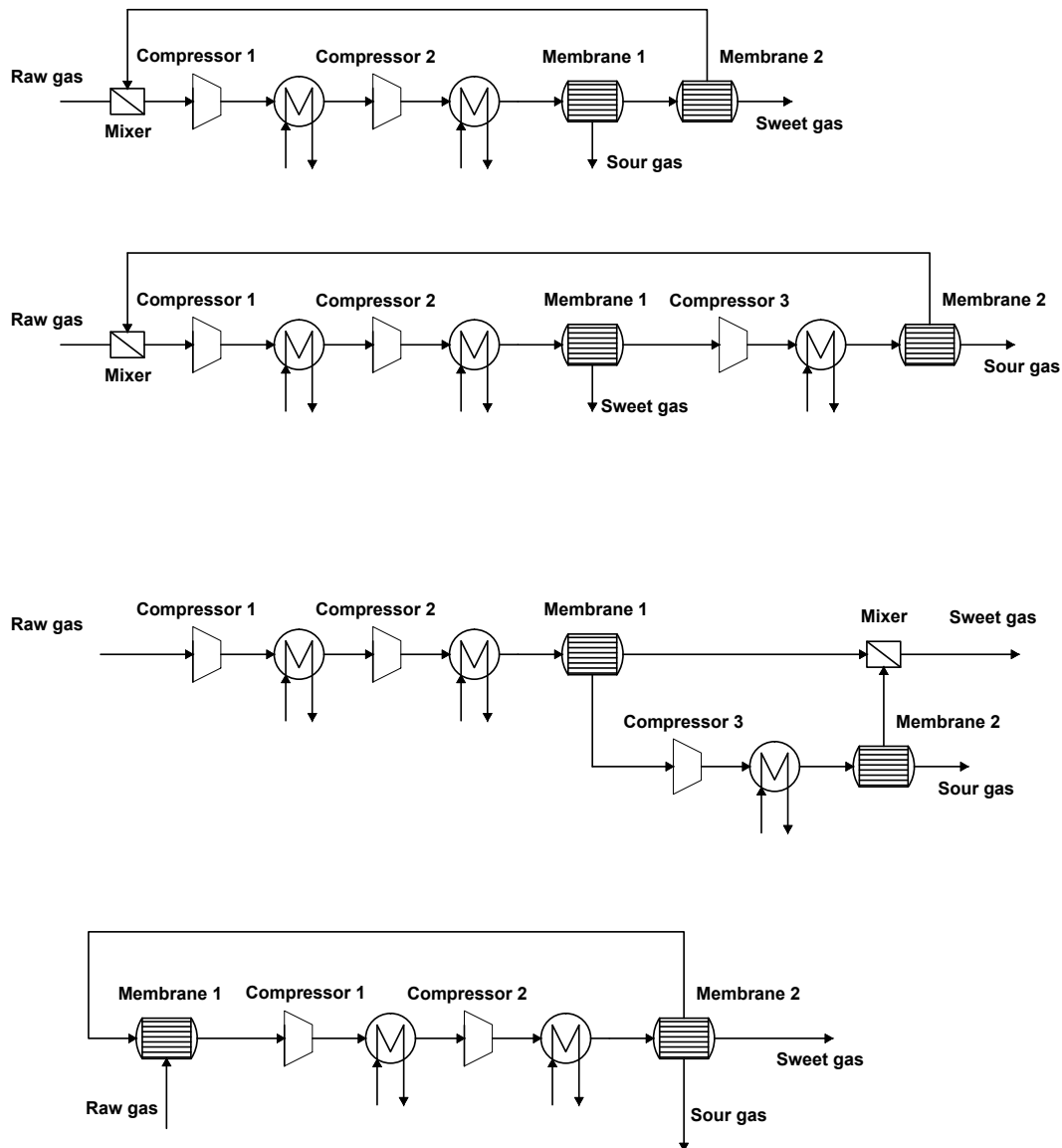


Figure 4. Multi-stage membrane system arrangements.

arrangements. A multi-stage configuration results in larger methane recovery and lower operational costs (Scholz et al., 2013). Among the multiple membrane stage systems, one can find configurations with one compression stage (Makaruk et al., 2010; Scholz et al., 2013) or multiple compression stages (Molino et al., 2013; Scholz et al., 2013). Some studies suggest that dual-stage membrane systems with a single compression stage can be the most flexible option, obtaining between 97% and 80% CO<sub>2</sub> recovery yields for CO<sub>2</sub> contents in the feed of 10–40% (mass) (Kim et al., 2017).

Since membrane materials are characterized by their gas permeability, many of these materials have been evaluated for CO<sub>2</sub> recovery, from traditional materials, including polymers, to the latest developed membrane materials, such as carbon molecular sieves. Among the common materials with larger CO<sub>2</sub> selectivity values, one can identify cellulose acetate, polyimide, and polycarbonate. Other membrane materials for the separation of CO<sub>2</sub> and CH<sub>4</sub> can be found in several reviews (Zhang et al., 2013; Chen et al., 2015; Vrbová and Ciahotny 2017).

### 2.3 Carbon recovery by chemical production

Several chemicals can be produced from the gases generated in the anaerobic digestion process. In particular, Hernández et al. (2017) developed a platform approach to produce dimethyl ether (DME), methanol, ethanol, and synthetic fuels based on the Fischer-Tropsch process. The production of hydrocarbons is based in the dry reforming of biogas to produce renewable syngas, which is their precursor. Due to the paramount importance of methanol, it was the first to be addressed (Hernández and Martín, 2016). In addition, DME, ethanol, and synthetic fuels can also be produced using the biosyngas as raw material by controlling the H<sub>2</sub> to CO ratio. Dry reforming is based in the equilibrium of the set of reactions shown in equation (1), which is dependent on the operating pressure and temperature. Dry reforming is an endothermic process, which can be heated by combusting a fraction of the generated raw biogas.



After the syngas production, methanol is produced following the set of reactions described in equation (2), catalyzed by a solid catalyst formed by a mixture of CuO–ZnO–AlO.



The production of methanol is favored at high H<sub>2</sub>/CO ratios, between 1.75 and 3, and it is considered that a CO<sub>2</sub> concentration between 2% and 8% is necessary due to the reaction mechanisms, see equation (2). Since the reaction equilibrium process is exothermic, and the concentration of species is lower at the right side of the reactions, the optimal operating conditions are defined by low temperatures and high pressures (Hernández and Martín, 2016). However, high pressure implies higher compression costs. Therefore, the trade-off between yield and operation cost is sought. The current industrial processes operate at pressures of 50–100 bar and temperatures of 473–573 K to activate the catalyst and boost the reaction (Hernández and Martín, 2016).

Furthermore, the production of ethanol and DME from biosyngas requires H<sub>2</sub>/CO ratios of 1. Regarding the DME production, this is produced via methanol as an intermediate product using copper catalysts, followed by a dehydration process catalyzed by acidic catalysts. As it is shown in the set of equations described by equation (3), together with the methanol and DME production, the water gas shift reaction takes place as well.





Similar to methanol production, DME production is governed by chemical equilibrium and is usually carried out at temperatures between 508 K and 553 K. A more detailed description of DME production from biosyngas can be found on Peral and Martin (2015).

The production of ethanol from biosyngas can be achieved following two paths. One is the chemical synthesis of alcohols using catalysts, producing ethanol and other alcohols as byproducts. This is an energy intensive process since the biosyngas must be heated up to 573 K and it operates at high pressure. As several alcohols are synthesized, a further purification process must be implemented. Another alternative for ethanol production is the fermentation of the biosyngas in a bioreactor. This biological process operates at low temperature, 311 K, and low pressure. However, the water-ethanol separation, which is energy intensive, must be carried out. Additionally, to ensure the survival of the biota of the bioreactor, the ethanol concentration must be kept below 5% by feeding water in order to maintain the ethanol concentration under this threshold, thus limiting the yield of the process. More details of these processes can be found in Martin and Grossmann (2011).

Finally, synthetic fuels can be obtained from biosyngas based on Fischer-Tropsch (FT) processes. The FT processes are based on reactions equivalent to polymerization reactions where the chains of hydrocarbons grow due to the dissociation of CO over a catalyst, as shown in equation (4), where  $nc$  denotes the number of carbon atoms and  $nh$  the number of hydrogen atoms. Usually, the catalysts used in FT processes are based on cobalt or iron.



Other reactions that take place in the process are the methanation of the  $\text{CO}_2$  and the water gas shift reaction, as described in equation (5).



FT synthesis can be carried out following two paths: High and low temperature processes. Both processes require  $\text{H}_2/\text{CO}$  ratios between 1 and 2. Each path is associated with a different distribution of products. On the FT high-temperature process, the fuel production is achieved at temperatures of 590–630 K and pressures between 10 and 40 bar, using iron-based catalysts. In this process, only traces of heavy fractions are produced, making the further separation processes easier (Hernandez and Martin, 2018). This separation process consists of a first separation of the light hydrocarbon compounds as gases, the removal of water generated in the process, and a distillation column to separate the different liquid alkanes. The low-temperature FT process is performed at 440–530 K with cobalt or iron-based catalysts. This process produces larger amounts of heavy fractions than the high-temperature FT process. Therefore, a hydro-cracking step must be included in order to break the long chain alkanes and enhance the conversion to gasoline and diesel. The rest of the separation step is similar to separation steps in the high-temperature FT process.

### 3. Carbon Footprint Mitigation by Nutrient Recovery

Anaerobic digestion processes generate a nutrient-rich residual stream, called digestate, which must be further processed in order to prevent soil and water contamination by nutrient pollution. Nutrient management is needed in order to prevent excessive releases of phosphorous and nitrogen to surface and



underground water bodies (eutrophication) and can lead to the exponential growth of algae, cyanobacteria, and the occurrence of harmful algal blooms (HABs) (Sampat et al., 2018; García-Serrano et al., 2009).

Some of the harmful effects of HABs over the environment are reflected in both water and air. Moreover, the overgrowth of algae and cyanobacteria shifts the distribution of aquatic species, leads the depletion of oxygen due to algal decay, and produces cyanotoxins that are harmful to aquatic life, wildlife, and humans. In addition, the development of HABs and eutrophication increases the emissions of  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , which are strong greenhouse gases (GHGs). On a 100-yr time horizon,  $\text{N}_2\text{O}$  and  $\text{CH}_4$  have 265 and 28 times the effect of  $\text{CO}_2$  on atmospheric warming, respectively (DelSontro et al., 2018). Currently, GHG emissions from lakes and impoundments are around 20% of global  $\text{CO}_2$ -equivalent attributed to fossil fuel emissions, with 75% of the impact due to  $\text{CH}_4$ . In addition, global increases in eutrophication could cause a 5–40% increase in the GHG effects in the atmosphere. This is the equivalent effect of a 13% global increase in fossil fuel combustion emissions (DelSontro et al., 2018). Beaulieu et al. (2019) predict the total phosphorus concentrations in lakes and impoundments to increase by up to 3 times by the year 2050, leading the development of eutrophication processes with an associated increase of  $\text{CH}_4$  emissions of 90%. Therefore, proper management of the nutrients contained in organic waste must be carried out in order to prevent the occurrence of eutrophication, HABs, and the subsequent damages to aquatic ecosystems and GHG emissions to the atmosphere.

However, when effective waste management practices are adopted, the nutrients in the digestate can be recovered as fertilizer, reducing the reliance on fossil-fuel based chemicals and the extraction of virgin feedstocks. There are several processes that can be used to recover the digestate nutrients. These range from simple mechanical separation, such as filters (Gustafsson et al., 2008), aerobic decomposition (U.S. Department of Agriculture (USDA), 2009), or chemical processing, such as struvite precipitation (Bhuiyan et al., 2008). The nutrient-rich material recovered in each case presents different characteristics affecting its final use, from a mixture of nutrients and organic matter to precipitates with a well-defined composition, as is the case of struvite production. The use of rich nutrient content products has been studied in the literature, showing the fertilizing efficiency of different materials (Hylander et al., 2006; Doyle and Parsons, 2002). Finally, it should be noted that quality variability in the recovered product, selling price, and production cost present complex trade-offs which should be considered at the decision-making step in order to select the most appropriate technology for each case (Martín-Hernández et al., 2018). There are a number of technologies that enable the recovery of nutrients from waste, among them we highlight the following:

### **3.1 Composting**

Composting is one of the simplest ways to manage solid organic waste, obtaining a valuable product which can be used as soil conditioner, and improving the soil fertility (USDA, 2009).

In short, composting is the aerobic biological decomposition of organic matter. For the decomposition of organic waste, particularly when the source of the residue is from livestock facilities, a pretreatment stage has to be implemented in order to reduce the moisture of the waste, since the raw waste has a high-water content which should be eliminated in order to enhance the microbiological growth. Therefore, a liquid-solid pretreatment must be implemented. Nevertheless, a significant amount of inorganic fraction of nutrients, which are water soluble, will be eliminated in the liquid fraction and will not take part in the composting process, posing an environmental threat due to nutrient pollution. On the other hand, composting converts nitrogen in ammonia form, which is volatile and can be eventually released to the atmosphere. In addition, composting must be complemented with external nutrient sources as it contains low amounts of nutrients in relation to its total mass. Other advantages of composting are the reduction of organic matter, greater ease of handling over raw waste, and pathogen elimination.

Composting can be carried out by implementing three methods. Windrow composting consists of making large windrows of organic matter to enable natural biological decomposition. The piles should be turned periodically in order to expose the material to the air. However, the exposition of the waste to air leads to odor issues. A second method is to make a pile of organic matter over a structure where the air is forced or drawn to aerate the organic matter. To enhance the decomposition and improve odor control,

the piles are usually covered. Finally, advanced compost systems run the decomposition in vessels where temperature, aeration, and moisture are controlled. However, the capital cost of these units is high and requires advanced management skills from the operators for its satisfactory performance.

The greatest disadvantages of composting methods for nutrient management is the variability of the final product. Since the ambient temperature, moisture and aeration conditions are not completely controlled, except in the case of high-cost sophisticated vessel composting systems, the recovered product characteristics are unpredictable (USDA, 2009).

### **3.2 Filtration**

Filtration is a low-cost technology which can be suitable for facilities where the recovery of a high-value product is not necessary. This technology can employ a filter containing a reactive medium that enhances phosphorus removal. Reactive filtration occurs through various mechanisms, depending on the characteristics of the filter media. For instance, filter media made of compounds rich in cations under basic conditions, usually containing calcium silicates at pH values above 9, promote the formation of orthophosphate precipitates in the form of calcium phosphates, mainly as hydroxyapatite (Pratt et al., 2012). Metallurgical slag as the filtration medium captures phosphorous by adsorption over metal at pH values close to 7 (Pratt et al., 2012). Package filters are considered for small facilities, while gravity filtration systems are more appropriate for large installations, according to recommendations reported by the U.S. Environmental Agency (EPA) (US EPA, 1979). These filters are used for phosphorous recovery in wastewater treatment facilities (Gustafsson et al., 2008) and further analysis can be found in Shilton et al. (2006), reaching phosphorous removal efficiencies up to 90%. Note that the recovered product is not a high purity chemical compound (unlike struvite), since the recovered solid compound is a mixture of nutrients and organic matter.

### **3.3 Coagulation**

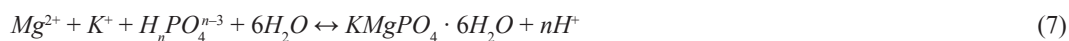
Coagulation is based on the destabilization of colloidal suspensions, achieved by reducing the attraction forces between phases. This is followed by a flocculation process, where flocs from the previously destabilized colloids are formed, raising their density, and resulting in a precipitation step which separates the precipitated solid phase from the fluid phase. Therefore, the nutrients, along with other sedimented solids, are recovered by clarification. Regarding the mechanisms followed by nitrogen and phosphorous in the coagulation-flocculation process, phosphorus is removed primarily in the form of complexes with metal hydroxides, which is the dominant process operating at typical pH values between 5.5 to 7 (Szabó et al., 2008). On the other hand, nitrogen recovery is related to the removal of the colloidal phase (Aguilar et al., 2002). Different coagulation agents used for the recovery of nutrients are  $\text{FeCl}_3$ ,  $\text{Fe}_2(\text{SO}_4)_3$ ,  $\text{Al}_2(\text{SO}_4)_3$ , and  $\text{AlCl}_3$ .

The removal efficiency achieved is similar for different coagulant agents, with values up to 99% for phosphorus and 57% for nitrogen (Aguilar et al., 2002). The coagulation-flocculation process performance is strongly influenced by the initial ratio of metal to phosphorus, pH, and chemical oxygen demand (COD). In particular, some previous studies report that the initial metal-phosphorus molar ratio must be between 1.5 and 2.0 and a recommended pH range of 5.5 to 7. However, a COD increment has a negative impact on the removal efficiency (Szabó et al., 2008). As in the previous method, the recovered product is a mixture of nutrients in an organic phase.

### **3.4 Struvite production**

Phosphorous and nitrogen can be recovered from the digestate through the formation of struvite, a mineral containing phosphate, ammonium, magnesium, and forming crystals with a chemical formula of  $\text{MgNH}_4\text{PO}_4 \cdot 6\text{H}_2\text{O}$ . The advantage of this technology is that the recovered struvite is a solid with well-known nutrient content, easy to transport, and can be used as slow-release fertilizer without any

post-processing step (Doyle and Parsons, 2002). The removal of nutrients via struvite production follows the reaction equation described in equation (6), by adding magnesium chloride ( $MgCl_2$ ) and resulting in the production of struvite crystals that can be recovered as solids. Due to the presence of potassium in the digestate, together with struvite, another product called potassium struvite or K-Struvite, can be produced, as described in equation (7), where the ammonia cation is substituted by the potassium cation (Wilsenach et al., 2007).



Some characteristics of livestock organic waste must be considered when designing a struvite production facility. Particularly, the high concentration of calcium ions that compete for phosphate ions and the presence of large amounts of suspended solids in the medium, which can interfere in the struvite formation and crystal growth step. However, some studies report successful struvite formation from livestock wastes (Schuiling and Andrade, 1999; Qureshi et al., 2008; Zhang et al., 2014; Zeng and Li, 2006).

At commercial scale, the advancement of the company Ostara®, which has developed commercial fluidized bed reactors for struvite formation in wastewater treatment plants (Ostara Nutrient Recovery Technologies Inc., 2019), is remarkable. This has opened a promising path to achieve technical and economic feasibility for struvite formation from agricultural residues, particularly regarding the processing of livestock waste.

## 4. Renewable Energy Incentives as Carbon Reduction Initiatives

In the context of carbon reduction initiatives, incentives to promote the installation of technologies to produce renewable energy are considered. Some studies describing the effects of incentives in the economic performance of these processes show how appropriate policies (in the form of incentives) can influence the decision-making procedure for accepting or rejecting the construction of facilities for bio-methane production, carbon reduction, and nutrient recovery.

### 4.1 European Union and United States

The European Commission established renewable energy targets of 20% final energy consumption from renewable sources by 2020 (Renewables Directive 2009/28/EC). This target enforces the European member states to reach their own renewable energy objectives, promoting the development and installation of renewable energy production facilities. In this context, the production of bio-methane plays an important role due to its flexibility in electricity and heat production, as a transportation biofuel, and in its storage. Additionally, given the decentralized nature of the farms, the installation of biogas facilities can contribute to rural community development. In addition, at the mid-term time scale, there are additional targets that must be matched by 2030 is 32% of energy demand should be covered by renewable energy. To achieve these goals, the European Union provides funding on a sub-national level (i.e., regions or municipalities) through instruments as the European Agricultural Fund for Rural Development (EAFRD) (Regulation (EU) No 1305/2013 of the European Parliament and of the Council, of 17 December 2013) and the European Regional Development Fund (ERDF) (Regulation (EU) No 1301/2013 of the European Parliament and of the Council of 17 December). Additionally, tax exemptions are applied in order to promote biofuel production under Directive 2003/96/EC (Council Directive 2003/96/EC of 27 October 2003). Beyond the European legislation, each member state can promote its own regulations to promote renewable energy production (European Commission, 2013).

In this context, there are four specific support schemes adopted in the European Union to promote the installation of organic waste treatment and the generation of renewable energy:

- The Feed-in tariff (FiT) scheme provides a remuneration per a certain amount of renewable energy produced. The value range of FiT varies across countries and the installed power capacity of the facility from 70 to 390 EUR/MWh (Deremince et al., 2017).
- Feed-in premium (FiP) are bonuses paid above the benchmark market price to subsidize the generation price of renewable energy. The value range of FiP varies across countries and the installed power capacity of the facility from 100 to 330 EUR/MWh (Deremince et al., 2017).
- Green Certificates (GC) are generated by the production of a unit of renewable energy and can be sold separately from the energy on the green certificates market, similarly to the Renewable Energy Credits implemented in the USA.
- Fiscal incentives are tax exemptions to make renewable energy more competitive in the market, as previously discussed (Deremince et al., 2017).

Additionally, another measure which would promote the reduction of carbon emissions from organic waste is the application of carbon credits to the emissions from organic waste. Carbon credits are a permit which allows the emission of a certain amount of GHGs. The amount of carbon emissions allowed for a certain sector is given by the number of permits emitted, and the GHG emissions allowed by each permit. These permits are tradable, as in the case of Green Credits, in a carbon market where companies can sell and buy the rights to emit GHGs. The goal of carbon credits and the carbon market is to control the GHG emissions through the purchase of a permit, which amount is limited. Therefore, following the market rules, their price will increase or decrease as a function of the GHG emission demand, and subsequently, the larger the emissions are the more amount of money the companies would have to pay. Other mechanisms for distributing carbon credits may be purchasing at auctions or awarding permits as offset for the implementation of clean technologies. Additionally, governments may distribute a certain number of permits at no charge, setting time-based emissions limits. As a result, only the facilities that exceed the limit set for a certain period of time must pay for additional permits, promoting compliance with the established standard.

Regarding the incentives available in the U.S., there are two main schemes: Renewable Energy Credits (RECs), and the Renewable Identification Numbers (RINs):

- Renewable Energy Credits is a mechanism created in the USA which guarantees that energy is generated from renewable sources, providing a system for trading produced renewable electricity. Each renewable megawatt-hour produced generates one REC that can be sold separately from the electricity commodity itself while representing the features of the generated renewable energy. RECs can be used to meet regulatory requirements by generators, trades, or end-users. The idea behind the implementation of RECs is to promote the use of renewable energy providing an incentive for its production by purchasing RECs similarly to the Green Certificates system developed in the European Union. The value of RECs fluctuate between 0.5 and 60 USD/MWh (Sampat et al., 2018).
- RIN is an identification number assigned to a batch of biofuel, allowing the tracking of its production, purchase, and final usage. Regarding the treatment of organic waste, the bio-methane produced through anaerobic digestion processes and its further upgrading can be eligible for the incentives associated to RINs in the form of liquified bio-methane under the Cellulosic RINs category (Sampat et al., 2018).

Another initiative that is available to promote the advancement and implementation of technologies for the management of carbon and nutrient from organic wastes is the Environmental Quality Incentives Program (EQIP) (USDA, 2019), managed by the Natural Resources Conservation Service (NRCS) of the U.S. Department of Agriculture. This program provides financial and technical support to agricultural producers so as to enable them to address improvements in their organic waste management installations and mitigate the environmental impact of their agricultural practices. The Rural Energy for America Program (REAP) (U.S. Department of Energy, 2019) provides loan financing and grant funding to agricultural producers so that they can install renewable energy systems. Anaerobic digestion facilities are eligible facilities among these programs (U.S. Department of Energy, 2019). Another incentive-based

program for public institutions was the (now repealed) Clean Renewable Energy Bonds (CREBs) for financing anaerobic digestion projects, among other technologies, giving federal tax credits, replacing a fraction of the usual bond interest, resulting in a lower final interest rate. Finally, the Interconnection Standards for Small Generators set the interconnection standards for generators under 20 MW to be connected with the utility lines (Federal Energy Regulatory Commission, Docket No. RM16-8-000; Order No. 828). These power generators, when installed in biogas facilities, qualify for these incentives.

It is challenging to compare the equivalent target for U.S. renewable energy production with the Horizon 2020 goals set in the European Union since the renewable energy standards in the USA are set at the state level. There are 29 U.S. states with established targets, 8 states with voluntary goals, and 13 states not having any renewable energy standards or target, as summarized in Table 1 (National Conference of State Legislatures, 2019). These renewable portfolio standards (RPS) require that a specified percentage of the electricity that utility companies sell comes from renewable resources.

In order to discuss the U.S. economic incentives associated with renewable energy and fuel recovery from livestock waste, three representative cases have been selected: A state with the most ambitious

**Table 1.** U.S. state renewable portfolio standards and goals (National Conference of State Legislatures, 2019).

State	Type of requirement	Requirement year	Renewable energy requirement (% or MW)	State	Type of requirement	Requirement year	Renewable energy requirement (% or MW)
AL	N/A	N/A	N/A	MT	Required	2015	15
AK	N/A	N/A	N/A	NE	N/A	N/A	N/A
AZ	Required	2025	15	NV	Required	2025	25
AR	N/A	N/A	N/A	NH	Required	2025	25.2
CA	Required	2045	100	NJ	Required	2030	50
CO	Required	2020	30	NM	Required	2020	20
CT	Required	2030	44	NY	Required	2030	50
DE	Required	2026	25	NC	Required	2025	12.5
FL	N/A	N/A	N/A	ND	Voluntary	2015	10
GA	N/A	N/A	N/A	OH	Required	2026	12.5
HI	Required	2045	100	OK	Voluntary	2015	15
ID	N/A	N/A	N/A	OR	Required	2050	50
IL	Required	2026	25	PA	Required	2021	18
IN	Voluntary	2025	15	RI	Required	2025	38.5
IA	Required	N/A	105 MW	SC	Voluntary	2021	2
KS	Voluntary	2020	20	SD	Voluntary	2015	10
KY	N/A	N/A	N/A	TN	N/A	N/A	N/A
LA	N/A	N/A	N/A	TX	Required	2025	10,000 MW
ME	Required	2017	40	UT	Voluntary	2025	20
MD	Required	2020	25	VT	Required	2032	75
MA	Required	2030	35	VA	Voluntary	2025	15
MI	Required	2025	35	WA	Required	2020	15
MN	Required	2025	26.5	WV	N/A	N/A	N/A
MS	N/A	N/A	N/A	WI	Required	2015	10
MO	Required	2021	15	WY	N/A	N/A	N/A



objective for renewable energy consumption, a state with intermediate goals, and a state with no established renewable energy requirements. As a measure of the determination to meet the different requirements, the fraction of energy that should be fulfilled from renewable sources and the type of obligation have been considered. Table 1 shows that the states of California and Hawaii present the most ambitious programs. For both states, their targets are to consume 100% renewable energy by 2045. However, California is selected as a more representative state because of its population, gross domestic product, and dimension. As an intermediate case, the average value of the renewable energy fraction covered from renewable sources was computed, obtaining a value of 29.7. The state with the value closest to this ratio is Colorado, where 30% of its energy demand should come from renewable sources. Finally, Alaska will be considered as representative of the states with no renewable energy targets (U.S. Department of Energy, 2019).

As a representative case of a determined renewable energy program, California has several incentive programs for the installation of anaerobic digestion technologies. Some of them consist of voluntary programs where the customers can support the development of renewable energy projects, such as the Pacific Power—Blue Sky Community Project Funds (Pacific Power, 2019). Furthermore, public funds are administered by the California Energy Commission (CEC), supporting renewable energy and research and development initiatives. Also, there are some feed-in tariff programs, such as the Marin Clean Energy—Feed-In Tariff, targeted towards small facilities (under 1 MW), or the Renewable Market Adjusting Tariff (ReMAT), available until the state cumulative capacity for the bioenergy ReMAT program reaches 250 MW. In addition, California offers a sales and use tax manufacturing exemption for electric power generation from sources other than a conventional power source and storage equipment under the “California Global Warming Solutions Act of 2006: Market-based compliance mechanisms: Fire prevention fees: Sales and use tax manufacturing exemption” (California Legislative Information, Assembly Bill 398 of 2017).

Regarding states with average renewable energy programs, Colorado offers incentives in the form of sales and use tax exemptions for renewable energy equipment across the state, as well as special tax incentives for economically distressed areas under the Colorado’s Enterprise Zone (EZ) program. For both programs, anaerobic digestion systems are eligible as renewable energy equipment. There are two additional incentives in the form of a regulatory policy which enforces municipal electric utilities serving more than 40,000 customers to offer an optional green-power program that allows retail customers the choice of supporting emerging renewable technologies, and through an interconnection standard for generators under 10 MW to be interconnected to intra-state utility lines.

Finally, the state of Alaska, which currently has no official renewable portfolio standards and goals, enacted the House Bill 306 in the 2009–2010 legislature, stating that 50% of the electricity produced should come from renewable energy sources by 2025. However, this House Bill was not enacted as a codified statute (U.S. Department of Energy, 2019).

None of these incentives consider nutrient pollution, neither in Europe nor in the USA. However, it is expected that the implementation of incentives towards avoiding nutrient pollution, in the form of phosphorous credits (P credits), can provide a mechanism to mitigate many environmental problems. P credits would work in a similar way to carbon credits implemented in the European Union, developing a waste management market around phosphorus. P credits would be related with the emission of a certain amount of P to the environment, penalizing the emissions, or in the opposite way, being an incentive for the recovery of phosphorous instead of being released to the environment. In both ways, P credits will be an incentive to promote the installation of organic waste management and nutrient recovery technologies.

A study about the effect of Renewable Energy Credits (REC), Renewable Identification Number (RIN), and P credits can be found in Sampat et al. (2018). The conclusions obtained in this work show that with the current system of incentives, the optimal strategy to manage waste is to deploy technologies that conduct simultaneous recovery of liquefied biomethane and nutrients in order to take advantage of RIN and P credits incentives simultaneously, obtaining a payback period of 4.8 years in the best case. Additionally, a report about the incentives and the current development of biogas facilities in the European Union can be found in Deremince et al. (2017).

## 4.2 Other countries

Carbon policies adopted by other countries are also based on carbon taxes, by setting a price for the carbon emissions, but without defining a cap for the emissions. Another scheme is the emissions trading systems (ETS) based on carbon credits, where, as in the case of European Green Certificates or the RECs implemented in the USA, there are permits allowing the emission of certain volume of GHGs. As detailed before, there is a market based on these permits, where companies can sell and buy the rights to emit GHGs. The following is a brief review of some national carbon pricing policies.

### 4.2.1 North America

- **Canada:** By 2019, Canada had adopted a carbon pricing system that introduced a federal regulation, allowing provinces and territories to adopt more restrictive policies or implement the federal administered system. This federal system, called output-based pricing system (OBPS), establishes a limit for emissions from each type of industry facility based on their annual economic production. Those facilities whose emissions exceed their annual limit must pay for additional emission permits. The rate for the additional permits is 10 CAD per equivalent ton of CO<sub>2</sub> emitted in 2018, rising to 10 CAN per year up to 50 CAD per equivalent ton of CO<sub>2</sub> emitted in 2022. Additionally, charges for GHG emitting fuels are considered. Four provinces, Quebec, Ontario, British Columbia and Alberta have their own carbon pricing system. Quebec and Ontario operate an integrated emissions trading system together with California for facilities emitting more than 25 kilotonnes of CO<sub>2</sub>eq per year, releasing auctions for the purchase of emission permits. British Columbia developed its own OBPS system in 2012, establishing an initial rate of 30 CAD per additional ton of CO<sub>2</sub>eq emitted. Since 2018, this rate will be raised by 5 CAN per year for four years. The agricultural sector is excluded from the OBPS system, while the liquefied natural gas sector is regulated under a specific OBPS system. Also, Alberta is regulated under its own OBPS system, charging 30 CAD per additional ton of CO<sub>2</sub>eq emitted. Only the facilities emitting more than 100 kilotonnes of CO<sub>2</sub>eq per year are subjected to the Alberta OBPS system. Saskatchewan, Manitoba, New Brunswick and Nova Scotia have announced their intention to implement their own carbon pricing system (Good, 2018).
- **Mexico:** In 2012, Mexico approved the Climate Change Law, taking the first steps towards the implantation of a voluntary emissions trading system. However, an amendment to this law approved in 2018 established that the Mexican ETS must be mandatory and adopted by 2022, after a pilot stage starting in 2019. The Ministry for the Environment and Natural Resources (SEMARNAT) is carrying out studies to set up the market rules for the pilot Mexican ETS. After the pilot-phase, the rules will be updated for the starting of the final emissions trading system (SEMARNAT, GIZ and German Federal Ministry for the Environment, Nature Conservation and Nuclear Safety, 2018).

### 4.2.2 Asia

- **China:** Pilot experiences for the implementation of emissions trading systems were implemented in the municipalities of Beijing, Chongqing, Shanghai, Tianjin and Shenzhen, and in the provinces of Guangdong and Hubei between 2013 and 2014, and in the province of Fujian in 2016. As a result of these experiences, in 2017, the National Development and Reform Commission (NDRC) of China announced the development of a national emissions trading system to be implemented by 2020. This system will cover the power generation industry at the beginning. The inclusion of other sectors in the ETS system has yet to be defined (Slater et al., 2018).
- **Japan:** Since 2010, the Tokyo Metropolitan Government launched a mandatory emissions trading system, with the objective of reducing CO<sub>2</sub>eq emissions by 25% to 27% from 2020 to 2024 and 30% by 2030. The estimated set for carbon price by 2018 is 5.89 USD per ton of CO<sub>2</sub>eq emitted (ICAP, 2019).
- **Korea:** In 2015, the Republic of Korea launched an emissions trading system called K-ETS. The Korean ETS system manages 69% of the national GHG emissions, establishing a GHG emission cap



of 538.5 megatonnes of CO<sub>2</sub>eq for 2018. The exchange of permits is implemented in the Emission Permits Exchange in the Korea Exchange (KRX) through auctioning of the allowances (The Government of the Republic of Korea, 2017; World Bank and Ecofys, 2018).

#### *4.2.3 Oceania*

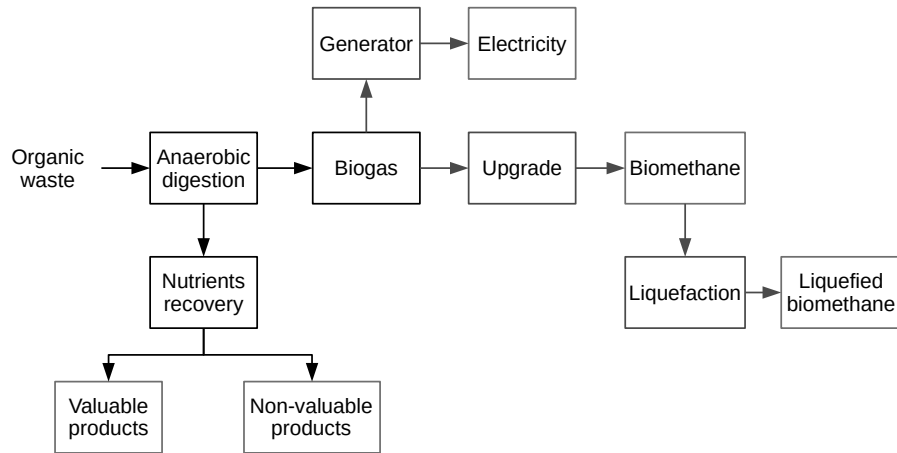
- Australia: Under the Emissions Reduction Fund and the Climate Solutions Fund, the Australian Government, through the Department of the Environment and Energy, promotes and supports the implementation of actions to improve the environmental sustainability of the stakeholders with the provision of 4.55 billion AUD. Among the eligible activities, there are actions to reduce the GHG emissions through the implementation of energy efficiency actions and waste management systems. The target is to reduce the emissions in 26%–28% below the 2005 emissions benchmark by 2030. Together with the government funding for the implementation of clean technologies and the deployment of measures to reduce the GHG emissions, Australia implemented an emissions trading system through the Australian carbon credit units (ACCUs). These permits can be sold to the Australian Government through the implementation of the mentioned actions to reduce the environmental impact of the stakeholders, or to other emitters who have to offset their emissions (Australian Government, Department of the Environment and Energy, 2019).

#### *4.2.4 South America*

- Argentina: Pursuing the discouragement of the consumption and production of GHGs emitting fuels, Argentina adopted a carbon tax for liquid fuels and coal in 2017, taxing 10 USD per ton of CO<sub>2</sub>eq emitted. It should be noted that natural gas is exempted of this carbon tax. This tax will be applied gradually from 2019 at 10% of the total rate, increasing 10% each year until reaching a 10 USD full rate by 2028 (Government of Argentina, 2018; FARN, 2017).
- Chile: Since 2017, a carbon tax has been implemented in Chile, taxing the CO<sub>2</sub> emissions at a rate of 5 USD per ton of CO<sub>2</sub>eq emitted. Only the facilities with a thermal power over 50 MW are subjected to the carbon tax (Government of Chile, 2019).
- Colombia: In 2017, a carbon tax was adopted by the Colombian government, including liquid and gas fossil fuels. The Ministry of Environment and Sustainable Development estimates that this tax will cover the 27% of the total national emissions. The rate imposed is 15,000 COP (approx. 5 USD) per ton of CO<sub>2</sub>eq emitted. However, the system contemplates exemptions from this tax for those stakeholders that implement measures to reduce GHG emissions or improve the energetic efficiency (MINAMBIENTE, 2017).

### **5. Effect of a Carbon Credit Policy Implementation in the USA: A Wisconsin Case Study**

We illustrate how carbon credits can potentially influence the management of the organic waste generated by concentrated animal feeding operations (CAFOs). We developed a case study for the treatment of organic waste produced at the 100 largest dairy CAFOs in the state of Wisconsin. This case of study was originally proposed by Sampat et al. (2018) to demonstrate the results of a proposed supply chain framework to evaluate the simultaneous effects of economic and environmental analysis of livestock waste. The treatment facilities considered include the installation of anaerobic digestion facilities, coupled with the production of energy from generators powered by the produced biogas, and the further upgrading of biogas to obtain biomethane or liquefied methane. The digestate produced in the anaerobic digestion process can be further treated for nutrient recovery in order to mitigate the eutrophication of water bodies and avoid the generation of GHGs associated with the presence of HABs. More details of the case study, including waste composition, the recovered product market values, and a list of candidate technologies with corresponding investment and operating costs can be found in Sampat et al. (2018).



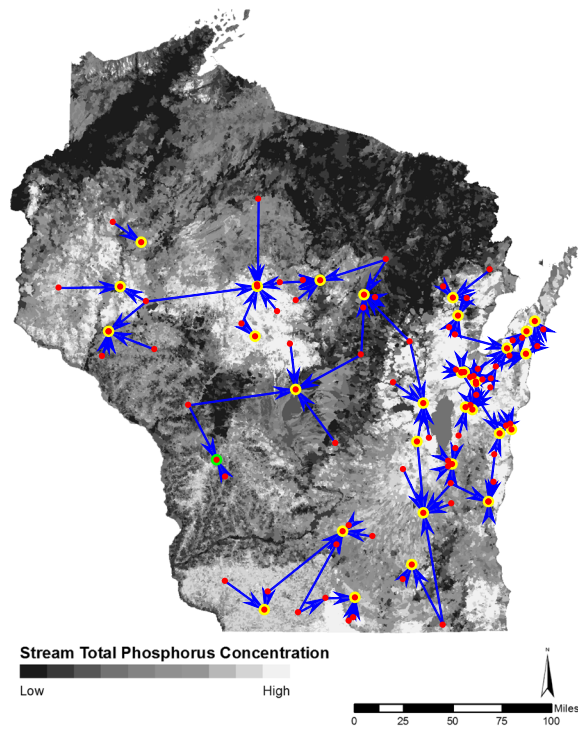
**Figure 5.** Possible configuration of the organic waste treatment facility considered in the case study.

The model determines the optimal configuration for each treatment facility that maximizes the total profit in the supply chain management network for livestock waste. Figure 5 shows the possible configuration of the waste treatment facility. The blocks outlined in black represent the common steps, a blue outline represents the different upgradation and refinement alternatives, and the red outlines denote the final products of the treatment process.

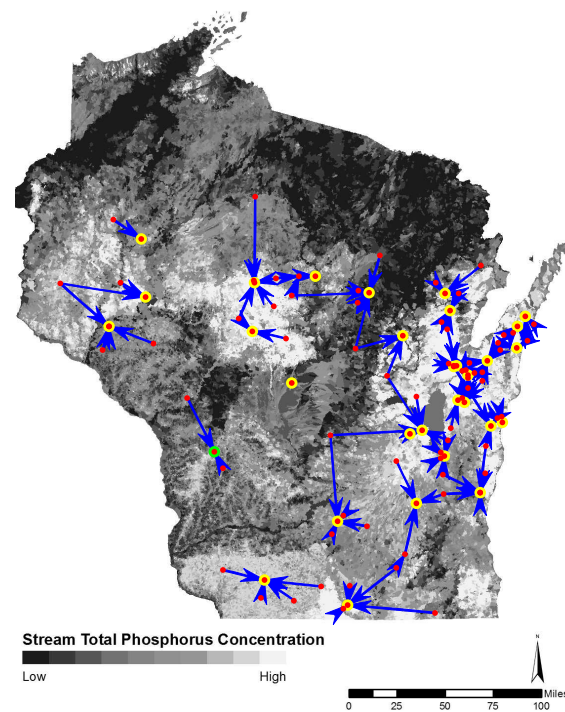
Sampat et al. (2018) show that the current product revenues obtained from the livestock waste treatment for nutrient and carbon recovery are not sufficient to make the system profitable. Therefore, in the base case scenario, they also consider the impact of economic benefits from the current U.S. incentives potentially available for these facilities, particularly RECs, RINs, and P credits. The values considered for these incentives are the following (Sampat et al., 2018):

- REC: 2 USD/MWh
- RIN: 2 USD/gal
- P credit: 22.04 USD/kg P

An alternative scenario proposed here would include carbon credits, assuming their current value in an existing reference carbon market, such as the European Union Emissions Trading System (European Commission, 2019b). This value is equal to 26.92 EUR per ton of CO<sub>2</sub> equivalent (Sandbag, 2019), which is equal to 0.11 USD per kg of C. The results from the base case scenario (considering REC, RIN, and P credits) are shown in Figure 6a, while the results from the alternative scenario when C credits are also included are shown in Figure 6b. The red dots indicate the location of CAFOs, yellow and green rings denote the placement of waste treatment facilities, blue arrows denote the transportation flows of the organic waste between the treatment facilities. Table 2 shows the results of both case studies. As a result of the implementation of the carbon credit policy, one can notice that, although the number of facilities for nutrient recovery facilities is the same (31), there is a change in the routes for the transport of waste reducing the transportation costs from 1.84 M USD/year to 1.80 M USD/year. Additionally, the location of the facilities installed is slightly different because of applying the carbon credit policy. Furthermore, additional revenue of 1.77 M USD per year is achieved, resulting in a slightly shorter payback period of 4.7 years in comparison with the payback period of 4.8 years if the carbon credits policy is not applied. The return of investment is larger with the application of the policy, 21.4% against 20.9% if the policy is not applied. It should be noticed that the application of the carbon credit incentives does not affect the type of technologies selected; for both scenarios, the production of liquefied methane and the recuperation of nutrients is selected because of RIN and P credit incentives. For both cases, RECs do not make electricity production economically sustainable.



**Figure 6a.** Base case scenario without considering carbon credits.



**Figure 6b.** Scenario considering the implementation of carbon credits policy.

**Figure 6.** Effects of the carbon credits policy implementation in the transport of manure in the state of Wisconsin (maps of the state of Wisconsin are adapted from U.S. Environmental Protection, 2014).

**Table 2.** Results of the case study considered with and without application of carbon credits policy.

	Without considering carbon credits	Considering carbon credits
Investment cost (USD)	4.36E+08	4.36E+08
Transportation cost (USD/year)	1.84E+07	1.80E+07
Operating cost (USD/year)	7.17E+07	7.17E+07
Revenue from products (USD/year)	3.78E+07	3.77E+07
Revenue from P credits (USD/year)	1.18E+08	1.18E+08
Revenue from C credits (USD/year)	0.00E+00	1.77E+06
Revenue from RECs (USD/year)	0	0
Revenue from RINs (USD/year)	4.75E+07	4.74E+07
Return on investment (%)	20.9	21.4
Payback period (years)	4.8	4.7
Nutrient recovery facilities installed	31	31

## 6. Conclusions

The release of carbon and nutrients from organic waste to the environment without proper management can result in serious threats to both the environment and human health. These threats are mainly related to the pollution of water bodies, and are associated with the expensive remediation treatments required in order to mitigate the adverse effects of pollution, and the release of GHGs to the atmosphere from the decomposition of the organic waste and algal blooms because of eutrophication processes. Therefore, adequate management of organic residues must be developed in order to make the production processes, particularly the livestock agricultural sector, one of the largest economic activities over the world, more sustainable, preserving the ecosystems and avoiding expensive remediation processes. In this chapter, several options for the management of organic waste, focusing specifically on carbon and nutrient recovery, have been described, with the aim of promoting their development and implementation in the current and future facilities in order to ensure the long-term economic and environmental sustainability of livestock operations. Additionally, the current U.S. and European Union incentives are described in order to give an overview about how governmental institutions around the world are promoting the installation of waste treatment technologies for energy and material recovery and developing the necessary infrastructure to reach a sustainable growth based around a circular economy. In this context, the development of circular economy waste management systems has a vital role in achieving sustainable use of raw materials and reducing the generation of residues. As it has been presented and following the premises of the waste to energy and chemicals concepts, the management and repurposing organic waste is a promising field for developing a circular economy system where valuable products can be recovered and reincorporated into the current productive loop.

Finally, this work discusses potential opportunities for the process design and logistics management of organic waste material under various types of government incentives. The consideration of these aspects would allow communities, investors, policymakers, and other stakeholders to conduct studies on the effect of economic incentives, environmental impacts, and technical feasibility of technologies for developing more sustainable carbon and nutrient recovery from organic waste.

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