Chapter 10
Positive Impact of Biogas Chain on GHG Reduction

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Abstract Nowadays, it is a well-accepted fact that greenhouse gases (GHG) contribute to the global warming of the planet and that they are a very real and very serious threat to the whole world. It is estimated that 10% of total GHG emitted is from sources in the agricultural sector and over 3% from waste management. Most countries agreed to reduce GHG emissions through the mitigation of GHG sources and application of technologies to stop global warming; however, there is much work to do as GHG are increasing every year. Among these technologies, anaerobic digestion appears as a well-established technology in most countries that can contribute to mitigate GHG emissions from organic wastes. Capture of these gases from uncontrolled organic wastes processes from municipal solid wastes, human excreta, wastewaters, tanneries, distilleries and other industries discharged in public swears is necessary to reduce these emissions and to profit methane from this biogas; otherwise, they are a source of fugitive GHG contributing to the global warming. Anaerobic digestion has the potential for global warming savings, due to the potential substitution of fossil fuel by biogas, also from carbon storage in soil and inorganic fertilizer substitution through use of the digestate as a fertilizer.

Keywords Global warming · Sustainability · Anaerobic digestion · Greenhouse gases · Organic wastes

10.1 Introduction

The United Nations Framework Convention on Climate Change (UNFCCC) is the main legal instrument for international response to the challenge of climate change that seeks to stabilize the concentrations of greenhouse gases (GHG) in the atmosphere to prevent dangerous anthropogenic disturbances in the climate system. To ensure the continuity of the efforts made with the Kyoto Protocol after 2020,

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the 195 member countries of the UNFCCC approved on December 2015 the Paris Agreement, which establishes measures to reduce GHG emissions through the mitigation, adaptation and resilience of ecosystems for the purpose of global warming.

According to the European Environment Agency, in 2015 total GHG emissions (excluding land use, land use change and forestry, in the EU-28 plus Iceland), amounted to 4317 million tonnes CO_2 equivalent (including indirect CO_2 emissions). Over 54% of the total was emitted from fuel combustion and fugitive emissions from fuels, over 24% from the transport sector, 8% from industrial processes and product use, 10% from sources in the agricultural sector (fuels and biomass burning, organic matter decomposition, soil tillage, etc.), and over 3% from waste management (Fig. 10.1) (Eurostat 2018). In the case of the energy sector, the most important energy-related gas is CO_2 that makes up 75%, followed by CH_4 that is responsible for 2% and N_2O for 1% of the total GHG emissions. Regarding the agricultural sector, contributions from CH_4, N_2O, and CO_2 of 242, 185 and 10.3 Mt CO_2-eq, respectively, represented 5.6, 4.3 and 0.24% of the GHG emissions, respectively. And finally, in the industrial processes and product use sector the most important GHGs are CO_2 (6% of total GHG emissions), HFCs (3%) and N_2O (0.3%) (EEA 2017). As indicated, the main GHG related to the agricultural sector are carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O). The effect of each on climate change depends on three main factors: concentration or abundance, residence time in the atmosphere and the impact strength in the atmosphere. For each GHG, a global warming potential (GWP) has been calculated to reflect how long it remains in the atmosphere and how strongly it absorbs energy. Gases with a higher GWP absorb more energy and thus contribute more to warming Earth. In spite of presenting the lowest GWP of 1, carbon dioxide has the highest direct warming impact because its concentration and the emitted quantities are much higher than that of the other gases. Methane is the second most important greenhouse gas, with a GWP of 23. Once emitted, methane remains in the atmosphere for approximately 9–15 years. Nitrous oxide is present in the atmosphere in extremely small amounts; however, its GWP is of 296 and has a very long atmospheric lifetime (114 years) (Steinfeld et al. 2006).

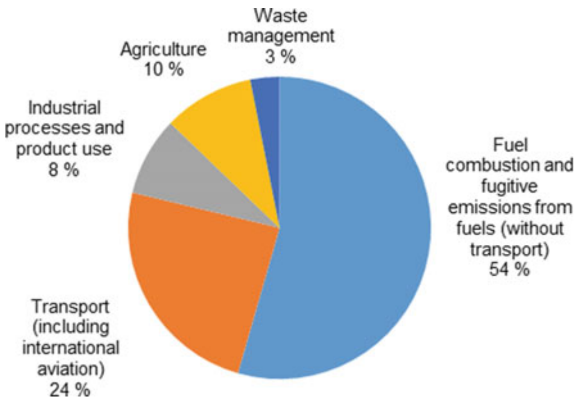

Fig. 10.1 Main contributors to GHG emissions in the EU-28. *Source* Eurostat (2018)

Livestock activities that produce large amounts of animal manure and slurries, as well as anthropogenic activities that produce wet and dry organic waste streams, emit considerable amounts of these three gases, and therefore, representing a constant pollution risk with a negative impact on the environment, human and animal health and food safety. To prevent emissions of GHG and leaching of nutrients and organic matter to the natural environment, it is necessary to close the loops from production to utilization by optimal recycling measures (Holm-Nielsen et al. 2009). One of these strategic measures is the application of the anaerobic digestion (AD) process to convert organic residues into energy and fertilizers, and therefore, to prevent GHG emissions.

According to different authors, AD contributes to GHG emissions, mainly from use of fossil energy at the facility, emissions from the bioreactor and combustion of biogas, and emissions from the digestate when applied to soil (Fig. 10.2) (Møller et al. 2009). However, AD also has the potential for global warming savings, especially from substitution of fossil fuel by biogas, also from carbon storage in soil and inorganic fertilizer substitution through use of the digestate as a fertilizer (Møller et al. 2009), and eliminating uncontrolled fugitive CH_4 emissions from stored wastes, as manure (Riaño and García-González 2015) or landfill (Yoshida et al. 2012).

In this chapter, the positive impact of the AD process on GHG emissions mitigation is described. The chapter is mainly focused on the treatment of organic wastes, which include animal manure and slurries, as well as wastewater and organic waste from municipal wastes. It describes the contribution of AD to emissions mitigation through renewable energy production, as well as different sources of fugitive GHG emissions related to organic waste degradation, how AD contributes to GHG emissions reduction compared with other technologies, some strategies to increase GHG mitigation during AD and, finally, the role of digestate management on GHG reduction.

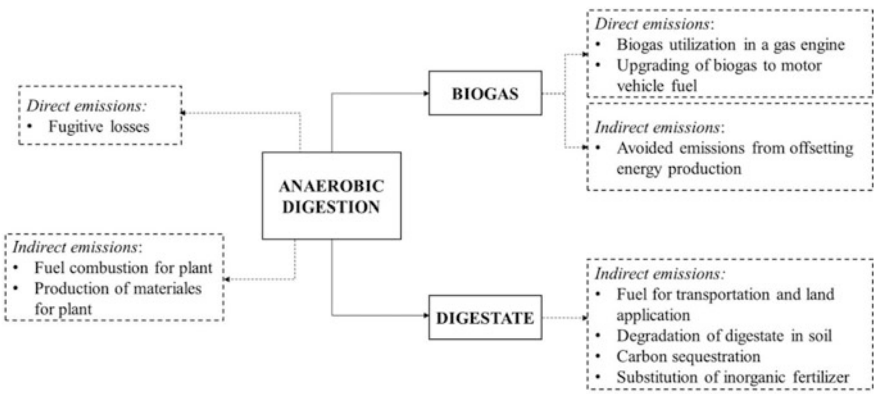

Fig. 10.2 Direct and indirect emissions from a biogas plant. Adapted from Møller et al. (2009)

10.2 Emission Mitigation Potential Through Renewable Energy Production

The need of GHG emissions mitigation is not only to create a sustainable environment but also to build a sustainable economy through renewable energy resources (IPCC 2014). As fuel combustion and fugitive emissions from fuels, as well as transport, play an important role as main contributors to GHG emissions, energetic alternatives must be developed and applied. In this sense, as transportation of people and goods play the most important role in CO_2 emissions, the EU has proposed a decrease in CO_2 emissions from vehicles of 30% by 2030 (European Commission 2017) by accelerating the uptake of zero and low emission vehicles (less than 50 g CO_2/km). The implication of governments and regional politics is essential to develop fossil-independent fuels like biogas. A successful example of the implementation of biogas technology for public transportation (mainly bus) has been performed in many cities, allowing the maturation of this technology (Ammenberg et al. 2018; IPCC 2014).

As mentioned, biogas from AD can be used for vehicles specially adapted to the use of methane, but a biogas upgrading step must be performed to separate methane from hydrogen sulphide, water and CO_2. Currently, biomethane is widely used for buses and heavy vehicles and different projects worldwide are performed to upgrade biogas and transform it in the main transportation biofuel (Holm-Nielsen et al. 2009; Ammenberg et al. 2018). From a theoretical point of view, biogas production may be tenfold higher than current production if food waste, agriculture, energy crops and industrial residues were utilized to produce biogas (IPCC 2014).

Also, hybrid vehicles equipped with both conventional (internal combustion) and electric engines may be an interesting alternative to decrease GHG emissions. However, most of the hybrid vehicles are based on petrol-fuels; hence, the use of a biogas-hybrid vehicle may result in an environmentally friendly alternative. On the other hand, electric vehicles do not emit GHG to the atmosphere and may be an interesting alternative to conventional cars, but currently an important amount of this electric energy is produced from fuel-based power plants. In this sense, a combination of non-fuel-based power plants and electric vehicles may be a suitable option to get zero-emission cars. Hence, there is a big potential for decreasing CO_2 emissions in the transport sector (European Commission 2017).

Biogas may be also used to produce electricity. In the EU, electricity from biogas increased from 3 GW in 2005 to 10 GW in 2015 (Scarlat et al. 2015). The main electricity producers from biogas are Germany, Italy and Czech Republic with 53, 13 and 4%, respectively (International Energy Agency 2016). Differences between them are remarkable, since Germany has favoured electricity production and biogas installation through positive policies, while supports in other EU countries are lower. In this manner, the main bottleneck for the electricity production from biogas in the EU is political issues (Scarlat et al. 2015). The reduction in support for biogas and the increase in costs to inject electricity into power grids have led to biogas producers to use the heat and electricity in their exploitations undersizing their biogas installations in order to avoid electricity surplus. However, despite the lack of tax incentives or bonuses, significant progress in installed electric capacity from biogas plants is expected as it is shown in Fig. 10.3 (Scarlat et al. 2015).

The worldwide share of biogas used as vehicular fuel is still very low (<1%) (Sahota et al. 2018), which is basically due to high operation costs and high energy consumption of the upgrading processes. In this sense, there is a need of establishing new biogas upgrading technologies and optimization of the older ones to reduce operation costs and consume less energy in the biogas creation (Sahota et al. 2018). Although in Europe, more than 90% of biogas produced is used for electricity generation, there is a tendency in biogas plants to upgrade their biogas instead of finding local sources for biogas consumption (Skovsgaard and Jensen 2018). The upgraded biomethane can be used as a substitute for natural gas, if the final composition meets the natural gas quality standards (Sahota et al. 2018). Therefore, the most promising future of biogas is to replace natural gas in its multiple uses.

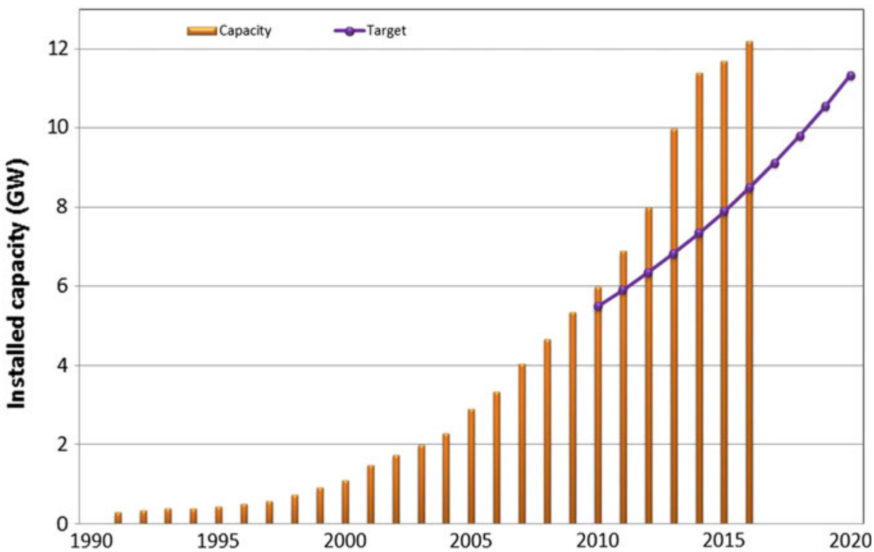

Fig. 10.3 Evolution of installed electricity from biogas plants (blue colour) and targets in the European Union (red colour). Adapted from Scarlat et al. (2018)

10.3 Fugitive GHG from Non-controlled Organic Waste Degradation

The main human-related sources of non-controlled GHG emissions are coal mining, production of natural gas and oil, livestock enteric fermentation, landfills, manure management, wastewater treatment and agriculture. Among these activities, only landfills, manure management and wastewater treatment will provide opportunities to reduce fugitive biogas emissions and to capture much of the generated biogas for its use as energy source (Abbasi et al. 2012), as the AD technology is already well-established in most countries. In this section, an overview of the fugitive GHG from landfills and manure management will be presented, as wastewater treatment is developed in Sects. 10.4 and 10.5.

10.3.1 Landfill

Landfilling is an important component of municipal waste management, a very common practice in many cities around the world. Methane and CO_2 are generated in landfills and open dumps as they are by-products of anaerobic decomposition of organic waste. The main components in landfilling are CH_4 (55–60%), carbon dioxide (40–45%) and N_2O (<5%) (Scheutz et al. 2009). Several factors influence methane generation in landfills. They include composition of the waste and availability of readily biodegradable organic matter, the age of the waste, moisture content, pH and temperature (Machado et al. 2009), as well as the design and management practices at the site (Abbasi et al. 2012). The processes that lead to the formation of landfill gas are bacterial decomposition, volatilization and chemical reactions.

Waste management is one of the main social and economic challenges, since it is estimated that production of municipal solid waste (MSW) in the world will double in next years, increasing from 1.32 billion tonnes per year in 2010 to 2.2 billion tonnes per year in 2025 (Pawlowska 2014). In 2014, 47.4% of the residues from EU-28 countries were deposited in landfills, being most of them open to the atmosphere, while 36.2% of wastes were recycled (Eurostat 2014). In the case of the USA, 54% of total municipal waste generated was discarded to landfill in the year 2012 (USEPA 2015). It is estimated that methane from the MSW landfills represents over 12% of total global CH_4 emissions (USEPA 2006). It amounts to over 730 million metric tonnes of CO_2 equivalent ($MtCO_2eq$). Countries like USA, Africa, Eastern Europe and China together account for 42% of the world's CH_4 emissions, and it is expected to reduce these emissions in next years due to specific regulations; however, in countries as India and Eastern Europe a steady growth in landfill CH_4 emissions is expected (Abbasi et al. 2012). This will be caused by the increase in municipal waste disposal, but also by the increasing proportion of biodegradable fraction of these wastes.

To control landfill emissions, three main strategies may be carried out: (1) To decrease the total amount of degradable waste left on the landfill; (2) to accelerate the AD process under controlled conditions (inoculation of anaerobic bacteria, temperature optimization, watering the landfill waste and diminishing oxygen concentration); (3) to collect and to burn the biogas produced. After biogas burning, emissions may be minimized by appropriate barriers. These practices will minimize the landfill impact on the atmosphere (Pawlowska 2014).

Collecting the biogas produced for its further use is the most spread strategy, avoiding its release from the landfills (Fig. 10.4). For that, perforated plastic pipes of about 15 cm in diameter are installed in the landfill. They are packed in gravel, and the pipe and the gravel are further enclosed in larger pipes. This is done to prevent refuse from plugging the perforations. A network of such extraction wells is installed across the landfill. Gas extraction can also be done by drilling boreholes in the landfill and installing extraction pipes. The individual gas wells are connected by a series of pipes leading to larger pipes that deliver the gas to the processing and conversion stations. The entire piping system is under a partial vacuum created by blowers or fans at the processing station, causing landfill gas to migrate towards the wells (Abbasi et al. 2012).

Although landfills are fugitive sources of GHG emissions, they still remain a good alternative for municipal waste disposal because of their simplicity and versatility, and as mentioned, there are strategies and technologies to control these emissions. More information about the control of GHG emissions from landfill sites can be found in Sect. 10.5. Recycling of non-renewable raw materials will prevent its disposal in landfills, as well as applying technologies to recover organic matter and nutrients from organic wastes.

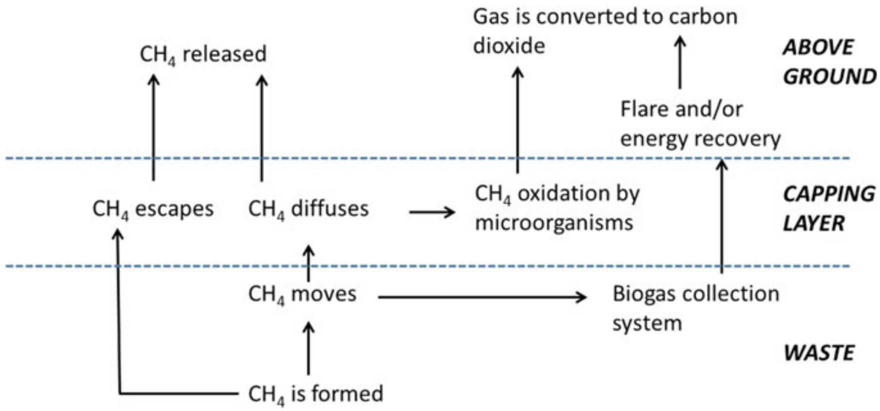

Fig. 10.4 Biogas release pathways in landfills. Adapted from Abbasi et al. (2012)

10.3.2 Manure Storage and Management

The handling and use of manure on livestock farms contribute to emissions of the GHG CH_4 and N_2O, especially with liquid manure management (Petersen 2018). GHG emissions from livestock vary by animal type and growth stage due to different diets, feed conversion mechanisms, while GHG emissions from storage and treatment of manure depend on the type of storage, duration of storage, ambient temperature and manure management practices (Chadwick et al. 2000; Borhan et al. 2012).

Most of the manure is collected in storage/treatment structures or left to decompose in the open, which poses a significant environmental hazard (Borhan et al. 2012), being therefore a fugitive emissions source. GHG emissions from animal houses are influenced by house ventilation, ambient temperature, floor type and existence and type of bedding material. In pig houses with natural ventilation, CH_4 emissions are significantly lower than in houses with forced ventilation due to lower temperature in the first ones. Concerning floor type, emissions from deep litter in pig houses are lower than those from pig houses with slats and slurry tanks (Sommer et al. 2013). Regarding N_2O, as manure remains in an anaerobic ambient there is little opportunity for the ammonia to be nitrified, and thus N_2O may theoretically be produced at the air–liquid interface of stored slurry or on slats and solid floors where urine and faeces are deposited. Emissions of N_2O may be affected by TAN concentration and pH, since high $NH_3(aq)$ concentrations inhibit nitrification (Sommer et al. 2013). However, in houses with deep litter systems N_2O emissions will vary depending on air exchange in the deep litter.

In outside slurry stores, CH_4 emissions vary over the year due to temperature variations and management practices (Sommer et al. 2009). In those countries where slurry stores are emptied in spring, only small amounts of slurry are exposed to high temperatures during summer, whereas in countries where slurry is stored in lagoons for years, emissions may be higher. Emissions may also be higher from lagoons that are not stirred, because dry matter settles out to the bottom and is seldom removed from the lagoons (Sommer et al. 2013). Usually, manure stored outside is under anaerobic conditions, thus emissions of N_2O via nitrification and denitrification without a floating cover are insignificant (Sommer et al. 2000). However, a natural or artificial surface crust on top of the stored manure can create anaerobic and aerobic conditions, thus creating an environment where N_2O can be produced (VanderZaag et al. 2009).

Depending on the management system employed to process manure, GHG emissions will differ significantly. Therefore, strategies for mitigating net GHG emissions should be aimed to manipulate manure properties or the conditions under which CH_4 and N_2O are produced and utilized during manure storage and treatment. However, GHG mitigation options are critical and depend on several factors, which are economic, technical and material resources, climatic conditions, existing manure management practices, bioenergy sources, and a source of high-quality fertilizer and soil amendments (Borhan et al. 2012).

According to Misselbrook et al. (2016), GHG mitigation techniques for slurry storage include promotion and capture of CH_4 in purpose-built anaerobic digestion plants, slurry crusting or covering the slurry surface with a floating material and slurry acidification. Methane produced by bacteria under stored conditions is transferred to the air above the slurry surface through ebullition. Therefore, surface crusts and coverings may provide the opportunity for CH_4 oxidation to CO_2, thereby reducing emissions (Husted 1994; Petersen et al. 2005; Qi et al. 2015; Sommer et al. 2000); as methanotrophs have been identified in slurry surface crusts (Duan et al. 2014). However, CH_4 can escape from the slurry through cracks or breaks in the covers, minimizing the oxidation of CH_4 (Petersen et al. 2013).

Manure management includes land application. In this case, emissions of CH_4 occur immediately after manure application to land and are usually short-lived, as oxygen diffusion into the manure inhibits CH_4 formation. Methane emitted immediately after application of manure to land is CH_4 trapped within the manure, having been generated during its storage (Sommer et al. 2013). After application of manure to soil, organic matter is mineralized forming ammonia that may be subjected to nitrification forming NO_3^-. Besides, O_2 demand increases and O_2 supply reduces in the soil, which affects the potential for N_2O emissions because the production is determined by the balance between O_2 demand and O_2 supply, rather than by O_2 supply alone. The effect of slurry will depend on soil conditions at the time of application. For example, in a dry soil increase in N_2O can be expected after slurry application, because slurry with a high content of degradable volatile solids increases O_2 demand and much more N_2O is produced. In this case, reducing degradable volatile solids content of manure through AD will reduce N_2O emissions (Sommer et al. 2013). More information on this subject is provided in Sects. 10.5 and 10.6.

10.4 Reduction of GHG During Anaerobic Digestion Compared with Other Technologies

Recent works about GHG emissions show that industrial and domestic wastewater treatment plants (WWTP) are anthropogenic GHG potential sources. Wastewater and organic wastes treatment can contribute to greenhouse gases through production of CH_4, CO_2 or N_2O from treatment processes as well as CO_2 produced from the energy required for treatment. Therefore, they contribute to the climate change and air pollution. The increasing interest towards climate change has led to the development of new tools for wastewater and organic wastes treatment plants design and management. Anaerobic digestion treatment plants are among these tools, which according to several studies have the potential for global warming savings.

In the last years, several studies have been conducted to compare GHG emissions from traditional wastewater treatment processes with those produced during anaerobic digestion. This technology results in a good alternative for the reduction of GHG produced during industrial and municipal waste and wastewater treatment. Although AD contributes to GHG emissions (see Sect. 10.5), this process also has a great potential for global warming savings. This section is devoted to briefly compile the reduction of GHG emissions during anaerobic digestion compared with other technologies or management practices applied to municipal, industrial and livestock wastes and wastewaters.

10.4.1 Agricultural Wastes

The constant growth of intensive pig farming has led to increased livestock waste in small and located areas worldwide. Within these areas, local use of manure as an organic fertilizer leads to nutrient over-application (N and P mainly) in agriculture, resulting in water and soil pollution (Bernet and Béline 2009). The most common management practice for liquid manure is to store it in uncovered anaerobic tanks for between four and six months, prior to exportation for landspreading (Burton and Turner 2003). Storing swine manure in uncovered anaerobic tanks entails a number of significant environmental impact issues, including GHG emissions (Riaño and García-González 2014; Vanotti et al. 2008). In fact, GHG released from livestock attributed to manure management account for 30% (Bernet and Béline 2009; Steinfeld et al. 2006). In this context, alternative technologies for manure treatment have been developed and implemented in order to achieve enhanced environmental protection, including the reduction of GHG emissions. These technologies include physical–chemical, aerobic and anaerobic processes.

Few comparative studies have explored the GHG emissions of the various manure management systems. Among those, some works compare GHG emissions of aerobic and anaerobic processes with the baseline scenario (i.e. conventional manure storage and further land application). For example, annual GHG emissions were cut by 62% through the installation of a swine manure treatment plant based on solid–liquid separation and nitrification–denitrification of the liquid phase compared with the baseline scenario (Riaño and García-González 2015). This reduction was in the range (53–75%) of estimated reduction for the implementation of anaerobic digestion for manure treatment (García-González et al. 2016). These authors calculated the GHG emission reduction for several full-scale anaerobic digestion plants that used manure as the main substrate, comparing with the baseline scenario. Most of the GHG emissions in these systems were produced in the final effluent storage ponds or in the intermediate manure storage before transportation in the case of collective treatment plants. In addition, methane leakages (estimated in 2% of the methane produced, according to the Swiss Quality Management Biogas Handbook) also had an important role in the GHG emissions in anaerobic systems. Collective treatment plants are sustainable, despite the higher GHG emissions due to transportation of substrates to the biogas plants. The GHG emissions’ reduction in anaerobic treatment plants is due, to a large extent, to the recovery of the biogas to produce heat and electrical power, avoiding the use of fossil fuels, and also to the very low methane production potential of the digested effluent. Differences in GHG emissions reported by García-González et al. (2016) for the different anaerobic digestion treatment plants were basically due to solid and nitrogen content in raw manure, as well as the differences in transportation distances from farms to the treatment plants. Another fact to consider is that as anaerobic digestion does not reduce N; the digestate will need further post-treatment or a land application planning. In this case, emissions derived from the fuel consumption during transport and land application of digestate will be the same as from raw manure. The digested product presents a lower dry matter content and thus a lower potential for CH_4 formation. However, when the digested manure had not been fully digested in the biogas plant, higher CH_4 emissions from digested than from untreated slurry during subsequent storage were observed (Clemens et al. 2006). Therefore, in anaerobic technologies, the hydraulic retention time must be long enough to exploit the potential for gas production without increasing GHG emissions during subsequent storage and field application (Clemens et al. 2006; Riaño and García-González 2015). A higher reduction percentage of GHG emissions (90%) has been calculated for manure composting systems (García-González et al. 2016).

In the case of dairy farms, several works have also evidenced the positive impact of the introduction of a biogas production plant. The storage of liquid manure for long periods of time without processing contributes to the most of GHG emissions during dairy manure management. The implementation of manure treatment technologies allows facilities to reduce emissions significantly, mostly through anaerobic digestion (Aguirre-Villegas and Larson 2017). Reduction of up to 50% of GHG emissions related to dairy manure management has been reported (Amon et al. 2006). In this case, the mitigation of GHG emissions is greatly due to the prevention of GHG emissions from undigested slurry storage (Battini et al. 2014). When producing electricity through anaerobic digestion, GHG emissions can be further reduced by replacing on-farm fossil fuel-based processed (Aguirre-Villegas et al. 2015). However, the mitigation of GHG is highly dependent on the fossil source to be replaced (Junior et al. 2015). In addition, it is important to point out that the poor management of on-farm digesters can compromise the environmental advantages of anaerobic digestion (Brunn et al. 2014). Thus, depending on the type of fossil fuel that is replaced by biogas, a poor management of biogas plants could led to the release to the atmosphere of between 3 and 51% of the biogas produced, which can have a great impact on global warming.

Despite the development of both aerobic and anaerobic manure treatment technologies, capital investment has been identified as the most important challenge facing implementation of cleaner treatment technologies, since these prove very costly compared to conventional manure practices (Vanotti et al. 2008). Fortunately, by adopting the Kyoto protocol, new programmes have been developed aimed at reducing anthropogenic emissions of GHG. Such programmes can help offset the higher installation costs of cleaner technologies and, therefore, stimulate their adoption by farmers (Vanotti et al. 2008).

10.4.2 Wastewater and Sewage Sludge

Municipal wastewater collection and treatment in wastewater treatment plants (WWTP) contribute to GHG emmisions due to biological degradation, being N_2O and CH_4 the main GHG contributors as it has been highlighted by the IPPC guidelines (IPCC 2014). According to Mannina et al. (2016), WWTP are one of the most important sources of anthropogenic CH_4 emissions, releasing close to 9% of total CH_4 emissions to the atmosphere. Regarding N_2O emissions, the US Environmental Protection Agency (USEPA 2006) estimated that WWTP are the sixth largest N_2O contributor; hence, these emissions must be controlled. Anaerobic digestion is one of the strategies to reduce these emissions. According to Wang et al. (2016), reductions of GHG emissions between 24 and 76% could be expected after utilization of biogas from anaerobic digestion of sludge in municipal WWTP.

Conventional processes for municipal wastewater treatment facilities, mainly based on nitrification and denitrification to remove nitrogen and organic matter simultaneously, are high energy and chemical intensive. Power is needed for running pumps and air blowers and for heating, and chemicals are mainly required for pH and alkalinity adjustment, phosphorous removal or other processes such as coagulation/flocculation. Electricity and chemicals have intrinsic carbon footprints, corresponding to the GHGs generated during their manufacturing and transport. In this way, the most significant contribution to GHG emissions in conventional processes for municipal wastewater treatment is these indirect emissions. In addition, the operation of WWTP results in direct emissions of GHG generated during the biological processes, such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O).

Technology innovation has a great potential for reducing energy consumption and GHG emissions from WWTP. A simultaneous reduction of energy consumption and an increment in energy recovery from wastewater would be important elements for achieving carbon neutrality (Wang et al. 2016). Caker and Stenstrom (2005) compared GHG production by aerobic and anaerobic treatment systems, including anaerobic wastewater treatment by processes such as the upflow anaerobic sludge blanket reactor and anaerobic filters. This study concluded that for very low strength wastewaters (less than 300 mg biochemical oxygen demand (BOD)/L), aerobic processes will emit less greenhouse gas. At higher strengths, anaerobic wastewater treatment would be more favourable, and the crossover point depends on the relative efficiency of the aerobic system. A technology to economically recover dissolved CH_4 from process effluents could make anaerobic wastewater more suitable in reducing GHG at all influents. Additionally, the combination of autotrophic processes to remove nitrogen, such as the partial nitrification and Anammox processes or microalgal-based technology, with anaerobic digestion could be an alternative. Using these technologies, oxygen requirements of aerobic processes are minimized while methane production is maximized (Campos et al. 2016).

Industrial wastewaters contain higher BOD and suspended solid (SS) concentrations than municipal wastewaters, leading to higher GHG production per m^3 of wastewater treated (Shahabadi et al. 2009). Taking into account GHG emissions, Shahabadi et al. (2009) recommended the combination of an aerobic reactor with anaerobic solid digestion with the recovery and use of the produced biogas for industrial wastewaters and specifically for food processing wastewater.

To minimize the GHG emissions in the WWTP, two main strategies may be implemented: (1) to prevent GHG emissions through a modification of the operation scheme in order to minimize the existing emissions, but this strategy may incur in important costs. (2) To use the current operation scheme of the WWTP and modify the operational conditions to decrease emissions and/or implement carbon capture and treat the gaseous streams. This strategy has remarkably lower impact than the first one (Parravicini et al. 2016).

10.4.3 Municipal Wastes

Disposal and treatment of municipal wastes are significant contributor to GHG emissions. The most extended system of municipal solid waste disposal is landfilling and that it is expected to increase due to the replacement of open dumping by landfilling in developing countries (Baldasano and Soriano 2000; Lou and Nair 2009). GHG emission of conventional landfills is highly dependent on waste composition, among other variables (Lou and Nair 2009). Studies suggest a large variation of GHG emission factors, varying between 1.3 and 2.0 t CO_2 eq per tonne of waste (Baldasano and Soriano 2000; Lou and Nair 2009). Landfill gas capture for flaring or combustion to recover energy is the most common mitigation strategy, showing a great potential for GHG reduction compared with conventional landfilling. Gas capture would led to a global warming potential (GWP) reduction of up to 58% of the total landfill's global warming potential (Liamsanguan and Gheewala 2008). Composting is considered a simple and effective way of treating the organic fraction of municipal wastes, while reducing GHG emissions. In this case, aerobic bacteria transform the organic matter to mostly CO_2 instead of CH_4, reducing the GWP of the landfill. Comparing with conventional landfills, a reduction of CH_4 concentration could achieve 90% in some cases (Cossu 2003; Lou and Nair 2009). In spite of this advantage, energy consumption associated with aeration is likely to be considerably higher than the operational requirement of conventional landfill. However, when the global GHG emissions are considered (decomposition and operational emissions), landfills appear to have a heavier impact on GHG emissions than composting (Lou and Nair 2009).

Anaerobic digestion presents advantage over composting, incineration or combination of digestion and composting mainly because of its improved energy balance (Edelmann et al. 2000). For instance, according to Liu et al. (2012) work, GHG reduction reaches 114 and 523 kg CO_2 eq per ton of waste for anaerobic digestion with power generation and bio natural gas compared with landfill baseline, respectively.

Finally, some authors have highlighted the importance of the implementation of an integrated municipal management system combining different treatments, as opposite to the use of one single process, for the reduction of GHG emissions (Baldasano and Soriano 2000). Thus, the calculated emission factor for landfilling is 1.97 t CO_2 eq per ton of waste, whereas the combination of sorting, wet biogasification, incineration and landfilling allows a reduction of 40% of GHG emissions. A combination approach is the best way to extract the material (e.g. recovery of nutrients as fertilizers) and energetic recycling potential of the different fractions of municipal wastes, to get the most out of these wastes.

10.5 Strategies to Increase GHG Mitigation During AD

Anaerobic organic matter degradation results in GHG formation. Carbon dioxide (CO_2) and methane (CH_4) are the main components of biogas, usually in a proportion of 25–50 and 50–75%, respectively (Wellinger et al. 2013). Methane is a valuable resource that can be easily converted into renewable energy but it is the second most prevalent GHG after CO_2. AD plants can be divided based on the substrate type into four categories (Deremince and Königsberger 2017). These are agricultural (energy crops, agricultural residues and catch crops), sewage (sewage sludge), landfill (biogas collected from organic waste disposal areas) and others (biowaste and municipal waste, household waste and industrial waste). Besides the potential GHG emissions from the AD plant itself, a variety of potential GHG emissions related to AD facilities have been identified (Fig. 10.5) (Burg et al. 2018). These include (1) waste disposal, (2) transportation to AD plant, (3) waste storage before AD, (4) AD plant, (5) conversion of biogas to energy and (6) digestate management before and during land application. In order to identify strategies to mitigate all the potential GHG emissions related to AD, this section reviews the potential sources of GHG emissions in AD facilities.

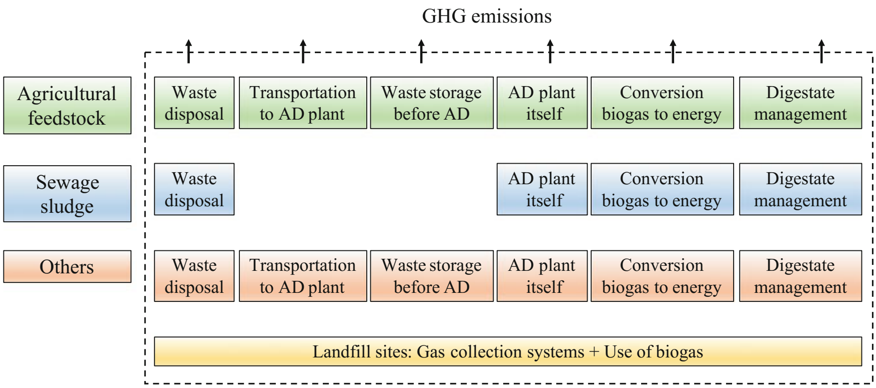

Fig. 10.5 Potential sources of GHG emissions in AD facilities. Adapted from Burg et al. (2018)

10.5.1 Agricultural AD Plants

GHG emissions in agricultural AD facilities can be divided into emissions from (1) waste disposal, (2) transportation of agricultural/livestock waste to AD plants, (3) agricultural/livestock waste storage before AD, (4) the AD plant itself, (5) conversion of biogas to energy and (6) digestate management. Manure is the main feedstock in agricultural AD plants, and most of the gas emissions in agricultural AD plants are related to manure disposal. These gases include CH_4, N_2O and CO_2. Methane production during manure management depends on the anaerobic conditions present in the farm. These emissions are higher if manure is treated as a liquid than if it is treated as a solid by-product. In the case of N_2O, it is released when the denitrification process is not completed. That is in the presence of anaerobic conditions, warm temperatures and carbon availability. The animal housing itself, more specifically littered systems, can be another source of N_2O. The Best Available Techniques (BAT) reference document for the intensive rearing of poultry or pigs compiles a wide variety of techniques that can be used to minimize GHG emissions during livestock waste management. These techniques include good housekeeping, nutritional management, efficient use of water and energy or on-farm manure processing, among others (Santonja et al. 2017). Carbon dioxide emissions related to transportation vary depending on the vehicle, but a standard emission factor of 0.43 kg CO_2/km can be applied to roughly calculate the emissions related to transportation (Burg et al. 2018). These emissions could be diminished by minimizing the distances from agricultural waste sources to AD facilities. The degradation of manure during storage prior to AD feeding could lead to both GHG emissions and potential energy losses. For instance, a study evaluating the CH_4 emissions pattern during pig slurry storage under Mediterranean conditions in summer recommended a maximum storage period of 30–35 days to prevent significant storage-related CH_4 emissions (Moset et al. 2012). Also the use of wooden lids placed on the slurry tank was found to reduce the net total GHG emissions in untreated cattle slurry and with anaerobically treated slurry (Clemens et al. 2006), and in general, some authors indicated that a solid cover or the presence of a surface crust on slurry stores reduced CH_4 emissions (Clemens et al. 2006; Husted 1994; Sommer et al. 2000).

The AD plant itself also accounts for the total GHG emissions. According to the Swiss Quality Management Biogas Handbook, a 2% of the annual amount of biogas produced can be assumed as emissions. In this vein, frequent controls should be done in order to identify biogas leakages (Liebetrau et al. 2017). A variety of available devices and methods for measuring emissions from AD plants as well as a review of different experiences in AD facilities is compiled in Liebetrau et al. (2017). Another potential source of emissions is the conversion of biogas to energy. Electrical energy and thermal energy are produced to ensure energy for the operation of the AD plant. The energy is produced by burning some of the produced CH_4, while CO_2 is released to the atmosphere. The use of this energy is not always optimized and up to 50% of the produced thermal energy sometimes goes to waste

(Szabó et al. 2014). Therefore, an effective utilization of the produced heat energy could be a strategy for GHG mitigation in this phase. Moreover, a control of the engine settings of the cogeneration unit should be regularly carried out to ensure complete combustion (Liebetrau et al. 2017). Finally, digestate management prior to field application is one of the main sources of GHG emissions. Two strategies are proposed to mitigate these emissions: (1) covering the digestate tank and (2) applying any aerobic post-treatment to avoid methanogenic activity (Liebetrau et al. 2017). Extended information about the role of digestate management on the reduction of GHG emissions can be found in Sect. 6.

10.5.2 Sewage Sludge AD Plants

Activated sludge treatment followed by AD of the sewage sludge in the same facility is among the most used technologies for sewage treatment nowadays. GHG emissions in sewage sludge AD facilities can be divided into emissions from (1) waste disposal, (2) the AD plant itself, (3) conversion of biogas to energy and (4) digestate management. Regarding waste disposal, GHG emissions are related to wastewater treatment and they include CH_4 and N_2O (Parravicini et al. 2016). N_2O is mainly produced in the activated sludge tank, as a product of the nitrification–denitrification processes. According to Parravicini et al. (2016), N_2O emissions from this tank account for a 26% of the estimated 36 kg CO_2 eq/year for a WWTP with AD. The optimization of the operational conditions of these processes is highlighted as the main strategy to reduce N_2O in WWTP-AD plants. Regarding CH_4, the sludge line in the WWTP is the main source of emissions before the sludge is fed to the AD plant. Reducing the time of sludge storage in tanks prior to AD would contribute to GHG mitigation. Regarding biogas conversion to energy, the efficiency of this process presents a high dependency on the country. For example, in Germany up to a quarter of the electricity and heat that is consumed in the WWPT is obtained from the produced biogas. On the contrary, in Brazil, most of the gas is nowadays burned and, therefore, not converted into bioenergy (dos Santos et al. 2016). In this country, the economic viability of electricity production from biogas depends on financial initiatives or consortia between neighbouring cities to build a centralized AD plant (dos Santos et al. 2016). AD is used as the last stabilization step for the primary and secondary streams obtained after activated sludge treatment of municipal wastewater. For this reason, the obtained product presents less risk of GHG emissions if compared to the digestate obtained in agricultural AD plants.

10.5.3 Landfills

GHG emissions from landfill sites include CH_4, CO_2 and small amounts of N_2O. A huge amount of biodegradable material ends up in landfill sites. This material is a potential source of CH_4 that is collected and mainly used for energy purposes. During the initial phase, landfill waste is not sealed and the biodegradable fraction experiences aerobic and anaerobic degradation. Consequently, CH_4 and CO_2 are emitted. Then, when the landfill is sealed, CH_4 production is increased and it is here where biogas can be collected, being this measure the main strategy to mitigate CH_4 emissions in landfill sites (EUROSTAT 2014). If the gas cannot be used to produce energy, it must be flared (Directive EC Waste Landfill 1999). The landfill directive includes some measures for landfill gas control and GHG emissions minimization, such as (1) lining of the landfill base and sides to create a low permeable barrier to subsurface gas flow or (2) surface sealing including impermeable mineral layers and gas drainage layers. Moreover, a variety of strategies has been proposed by the European Commission in order to maximize biogas recovery while minimizing CH_4 emissions. These strategies include (1) starting biogas collection as soon as possible, right after the deposit of the waste, (2) minimizing the area of waste not sealed, (3) installing gas collection systems as soon as possible, (4) sealing all landfill infrastructure such as leachate or gas wells to prevent gas leaks, (5) regular monitoring for all sealing systems to detect possible leaks and (6) regular maintenance and optimization of collecting systems. After biogas collection, this should be utilized and the maximum amount of energy should be obtained. Some techniques to optimize biogas utilization from landfill sites are (1) introduction of the treated CH_4 into the gas mains, (2) combined heat and power utilization, (3) direct use of the gas as a fuel or (4) electricity generation from biogas.

10.5.4 Other AD Plants

Other AD plants include biogas plants treating municipal organic waste, household waste and industrial organic waste. In many cases, organic household and industrial wastes are co-digested with manure in manure-based biogas plants. The potential GHG emissions sources are similar to those in agricultural AD plants: (1) waste disposal, (2) transportation of waste to AD plants, (3) waste storage before AD, (4) the AD plant itself, (5) conversion of biogas to energy and (6) digestate management. GHG emissions related to organic household disposal include the collection system, the frequency and the waste composition. An optimization of the separation and collection of organic materials in origin together with a selection of types of waste collected are proposed for reducing GHG emissions (Yoshida et al. 2012). In the case of organic industrial waste, the emissions due to the collection are reduced. Emissions related to transportation (CO_2) are mainly dependent on the vehicle, and they could be diminished by minimizing the distances from the waste sources to AD facilities. Potential emissions related to waste storage before AD, the AD plant itself and conversion of biogas to energy, as well as the strategies to reduce these emissions, are similar to those in agricultural AD plants. In this case, the use of this digestate as fertilizer is subjected to its content of pollutants and its application is ruled by each country legislation (Al Seadi and Lukehurst 2012).

As a summary, the main potential sources of GHG emissions in AD facilities are identified. Organic waste storage (including manure, digestate, organic household waste) is the main source of GHG emissions in agricultural and industrial AD facilities. Covering the storage tanks could be a strategy to mitigate those emissions. In the case of WWTPs, the main source of GHG emissions lies on the activated sludge tank, being the optimization of the operational conditions the main strategy to counteract those emissions. In the case of landfill sites, the spontaneous anaerobic fermentation of the disposed organic matter results in CH_4 and CO_2 emissions. To reduce these emissions, the collection of this CH_4 is of major importance, as well as to produce energy or further upgrade it to be used in vehicles.

10.6 The Role of Digestate Management on the Reduction of GHG

Digestate contains a high amount of organic matter and nutrients. Its use as organic fertilizer is gaining great interest day by day due to its economic and environmental advantages. Among others, these benefits include the energy and GHG emission savings, if compared to the production of inorganic fertilizers. However, there are GHG emissions related to management of digestate that should be evaluated. These indirect emissions are mainly produced during the storage, transportation and land application and soil degradation of the digestate. In this sense, the quality of the digestate is of major importance for its application as a fertilizer; since, for example, the most mineralized the digestate the less N_2O emissions will generate, as less degradable volatile solids in the digestate will decrease O_2 demand in the soil and therefore less N_2O will be emitted. Other characteristics of digestate are also important as specific chemical composition (i.e. content of nutrients, moisture, pH); safety standards according to the current legislation for each country (including pathogens, heavy metals and organic pollutants). In this vein, and due to a high risk of chemical contamination, digested sewage sludge or digestate obtained from industrial feedstock is only allowed to be used as a fertilizer in some European countries (Al Seadi and Lukehurst 2012). The present section briefly describes potential GHG emissions in these different steps.

10.6.1 GHG Emissions During Digestate Storage at the AD Plant

The main use of digestate is agricultural application as fertilizer. Since digestate is produced during the whole year and fertilization must be done during the growing season, a storage tank in the AD facility is needed. Alternatively, a storage tank for digestate can be placed close to the fields where it will be applied. Generally, the digestate is stored in uncovered tanks for up to 180 days from which GHG, such as CO_2 and CH_4, are emitted to the atmosphere (Menardo et al. 2011). Moreover, it can be dewatered to separate solid and liquid fractions for its easy handling and transportation (Zeshan and Visvanathan 2014). N_2O emissions during digestate storage are not expected to be a significant source of the total N_2O emissions from biogas plants, as the anaerobic conditions in the tanks prevent its production (Holly et al. 2017). However, digestate storage is a great contributor to CH_4 emissions from anaerobic systems. CH_4 emissions from digestate are not well quantified, with only few studies providing data. For instance, Baldé et al. (2016) estimated that annual emissions from earthen digestate storage were about 12% of CH_4 produced within the digester, thus counteracting GHG emission reductions that are usually assigned to AD. Gioelli et al. (2011) reported that the digestate storage accounts for about 27% of total CO_2-eq emissions generated during anaerobic processes. In spite of these high emissions, CH_4 emissions from digestate storage are substantially lower compared to untreated manure storage. Specifically, a reduction of 85% was obtained by Maldaner et al. (2018), when comparing total annual CH_4 emissions from untreated manure with the digestate tank at the same farm (accounting for 6.6 and 1.0 kg m^{-3} y^{-1}, respectively). Amon et al. (2006) also found that anaerobic digestion reduced GHG emissions by 60% from the untreated slurry due to the reductions of CH_4 emissions. Zeshan and Visvanathan (2014) calculated a decrease of about 75% in the GHG emission potential of digestate compared with organic fraction of municipal solid waste.

The reduction in CH_4 emissions from the digestate tanks is not only related to the degradation of part of the organic matter, but also due to an increase in the less digestible form of organic matter in the digestate (Maldaner et al. 2018). Indeed, the digestate organic matter content and its quality greatly affect to CH_4 emissions during storage. Both the amount and the quality of the organic matter are influenced by the technical and operating parameters of the biogas plant. Thus, in biogas plants operating at high organic loading rates (OLR) and at short hydraulic retention times (HRT), the digestate still contains a considerable amount of undigested organic matter that it is gradually digested during storage. Under such conditions, and if the storage tank is uncovered, a considerable amount of CH_4 could be released to the atmosphere. In these cases, the collection of CH_4 during digestate storage could be economically viable operating at high OLR and at low HRT. Besides, covering storage tanks offers an opportunity to reduce GHG emissions to the atmosphere while capturing residual digestate methane (Kaparaju and Rintala 2006; Menardo et al. 2011). On the contrary, the operation at low OLR and at very long HRT results in negligible emissions from digestate (Menardo et al. 2011).

Other factors encourage enhanced CH_4 production after AD. Specifically, the sludge layer in the storage tank is a key contributor to CH_4 emissions, since it could contain as much volatile solids as the annual discharge from the anaerobic digester (Baldé et al. 2016). Another determining factor controlling CH_4 emissions during digestate storage is the environmental temperature and that of the digestate entering the storage tank (Clemens et al. 2006; Sommer et al. 2007; Maldaner et al. 2018; Menardo et al. 2011). For instance, CH_4 emissions during summer were approximately 50% higher than during winter for biodigesters fed with a mixture of manure and crops (Liebetrau et al. 2013). Likewise, Rodhe et al. (2015) found negligible emissions from digestate in the winter. In addition, it is noteworthy that due to heating during anaerobic digestion, the high digestate temperature can enhance CH_4 emissions.

Minimizing the retention time during storage will reduce GHG emissions, since CH_4 emissions will be avoided. In addition, during storage part of the ammonia is volatilized due to favourable conditions of pH. The losses of N would decrease the potential value of digestate as fertilizer hence reducing GHG savings from fertilizer substitution by digestate (Zeshan and Visvanathan 2014). Digestate solid–liquid separation also affects to GHG emissions. Particularly, the effect of solid–liquid separation would depend on the type of biogas feedstock. Thus, Holly et al. (2017) found that a solid–liquid separation following anaerobic digestion reduced 68% of CH_4 emissions in digestate storage, compared with raw dairy manure storage. Perazzolo et al. (2015) observed that mechanical separation of anaerobically digested cattle slurries reduced GHG emissions by 40%, while on digested pig slurries no significant effect was observed.

As an overall, the operation of anaerobic digestion plants as well as digestate management is of key importance for minimizing CH_4 during digestate storage in biogas plants. Some potential best practices can be adopted for reducing GHG emissions including regular storage emptying, digestate solid–liquid separation and storage tank covering (Baldé et al. 2016).

10.6.2 GHG Emissions from Digestate During Transportation and Land Application

Emissions during digestate transportation and land application are highly dependent on the distance to the fields from the AD plant and if the digestate is further treated to reduce water content (Møller et al. 2009). This is due to the low dry matter content of digestate (<10%), that often makes storage and transportation expensive. Møller et al. (2009) have proposed a global warming factor in the range of 0.9–1.9 kg CO_2-eq per tonne of wet waste and 1.5 kg CO_2-eq per tonne of wet waste for transportation and land application of the digestate.

After land application, the biodegradation of the digestate begins, resulting in CO_2 and N_2O emissions. Emission coefficients for CO_2–C and N_2O–N are in the range 0.86–0.96 of the C and 0.013–0.017 of the N applied to the soil, respectively, depending on environmental conditions, soil characteristics and other parameters related to agriculture, such as the application technique (Møller et al. 2009). Comparative studies have evaluated GHG emissions from land application of digested and undigested manure; however, obtained data are very variable. Clemens et al. (2006) compared GHG emissions from untreated and anaerobically digested cattle slurry after land application, and they concluded that there were no significant differences between both types of slurry with annual emissions of 4.15–8.1 kg CO_2 eq. per m^3. Indeed, they found that GHG emissions from slurry storage are more important than emissions after field application. On the contrary, some authors report between 17 and 71% lower N_2O emissions from land application of digestate, compared with that from undigested manure, depending among others on the soil characteristics (Börjesson and Berglund 2007; Chantigny et al. 2007). This reduction has been attributed to the lower content in easily degradable C in digested feedstock, hence less energy source for denitrifier bacteria (Nkoa 2014; Vallejo et al. 2006; Rochette et al. 2000). Moreover, several works have concluded that digestate presents a higher risk of N_2O emissions than undigested manure. This higher risk could be due to the higher ammonium content of the digestate. Thus, Thomas and Hao (2017) indicated that N_2O emissions from soil receiving digestate were 4.3 and 3.6 times higher than the emissions of the separated solids and cattle manure, respectively. In addition, other studies have found higher N_2O emissions from soil amended with digestate than those from a soil amended with inorganic fertilizers. For example, Pampillón-González et al. (2017) evaluated GHG emissions during the growth of wheat cultivated in soil amended with digestate and concluded that although emissions of CO_2 and CH_4 were not significantly affected by fertilization, cumulative N_2O emissions increased by five times compared to urea-amended soil. The variability in results found in the literature highlights the importance of conducting additional research that explores the GHG emissions after digestate land application.

Several management practices can be adopted in order to minimize GHG emissions from land application of digestate. Spring application would mitigate N_2O emissions via the reduction of the amount of substrates necessary for the accomplishment of N_2O-related freeze–thaw processes (Nkoa 2014). Agricultural practices that enhance soil aeration and a good drainage would also mitigate N_2O emissions after the application of anaerobic digestates (Nkoa 2014).

Land application of digestate also would replace the use of inorganic fertilizers and, consequently, the GHG emissions from fertilizer manufacturing would be avoided (Pampillón-González et al. 2017). The average emission values for the production of nitrogen, phosphorous and potassium fertilizer calculated from Boldrin et al. (2009) are 8.9 kg CO_2-eq/kg N, 1.8 kg CO_2-eq/kg P and 0.96 kg CO_2-eq/kg K. The substitution of inorganic fertilizer will depend on the concentration and availability of nutrients in the digestate; therefore, it is important to characterize the composition of digestate before its use. Some other advantages that can lead to GHG reductions are the increment of water retention in the soil (thus reducing irrigation), the reduction of the requirement of herbicides or biocides, the improvement of soil structure or the reduction of the erosion (Møller et al. 2009). These savings are not yet well quantified; however, it is worth noticing that these induced effects on soil would address to important benefits for global warming. In any case, the use of digestate as a fertilizer for land application implies a high reduction of GHG emissions when compared with a scenario in which digestate is disposed in a dumpsite (Zeshan and Visvanathan 2014).

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