Worldwide Survey of Biodegradable Feedstocks, Waste-to-Energy Technologies, and Adoption of Technologies

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Abstract The current chapter survey categories of biodegradable waste, including manure and animal waste, food waste, crop residues, and sewage waste. The chapter then identify and analyze several major types of waste management technologies, such as anaerobic digestion, landfilling, composting, and incineration. It concludes with a brief discussion on the different patterns of adoption among regions.

1 Introduction

The generation of waste is an unavoidable consequence of most human activity. During the economic life cycle of the production and consumption of goods and services, undesirables are discarded from start to finish. These undesirables, if not managed properly, pose a serious environmental and public health risk. One category of waste – biodegradable waste – originates from plant or animal sources. Despite emanating from organic sources, improper handling of such waste can threaten air, water, and land resources. Globally, as both population and affluence increase, production of biodegradable waste is also likely to increase. Properly managing such waste as to reduce pollution and limit exposure while promoting nutrient recycling will be an important undertaking for both developed and developing countries in the years to come.

This book chapter has three main purposes: 1) to survey six broad categories of biodegradable waste; 2) to identify and analyze four major types of waste management technologies; and 3) to examine global adoption of such technologies. The six

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categories of waste (from here on, ‘waste’ will refer to only biodegradable waste) to be appraised are manure and animal waste, food waste, crop residues, sewage sludge, and a general category of other industrial, municipal, and residential waste. Next, the four technologies to be studied are landfilling, composting, incineration, and anaerobic digestion. Afterwards, we conclude with discussion of international adoption of such technologies.

2 Survey of Biodegradable Feedstocks

The waste categories surveyed below– manure and animal waste, food waste, crop residues, sewage sludge, and other waste –differ in terms of generation rates, worldwide distribution, and biological characteristics.

2.1 Manure and Animal Waste

Most animals, which are produced for human consumption, are produced in concentrated animal feeding operations or CAFOs (Burkholder et al. 2007). According to a 2004 EPA study, “One animal facility with a large population of animals can easily equal a small city in terms of waste production,” and a dairy farm with 2,500 cattle produces a similar amount of waste to a city of 411,000 people (EPA 2004). The abundance of CAFOs today presents challenges for animal waste management. The traditional method of manure management is land deposition, which simply involves dispersing the manure onto the fields as fertilizer, either in a solid or liquid form. This method is adequate for traditional farms that have small concentrations of livestock, but because of the size and concentration of today’s farms, large quantities of waste are produced in a relatively small area. The great volumes of waste produced often exceed the assimilative capacity of the land for any land within an economically feasible transport distance (EPA 2004). 133 million dry tons of manure is produced in the United States each year, and there is a question of how to effectively treat and dispose of this waste. Animal manure is high in organics, nitrogen, and phosphorus, which make it useful as a fertilizer or conditioner of farmland – however, if manure is over-applied to the land, the soil will be overloaded with nutrients and heavy metals (Burkholder et al. 2007).

There is a number of other environmental, public health, and water quality issues related to the treatment of manure from CAFOs. In the US, human waste is treated before disposal into the environment, but animal waste is not treated or minimally treated before disposal, such as in land deposition (EPA 2004). There are possible environmental contaminants in animal waste from CAFOs, including: nutrients, pathogens, veterinary pharmaceuticals, heavy metals (especially zinc and copper), and naturally excreted hormones. There are also over 100 microbial pathogens in swine waste that can cause human illness and disease. Parasites, viruses, and bacteria are present in animal wastes at the level of 1 billion per gram. Animal waste is also high in biochemical oxygen-demanding materials (BODs)—namely, in the quantity of oxygen used by microorganisms (e.g., aerobic bacteria) in the oxidation of organic matter. Swine waste slurry contains 20,000-30,000 mg of BODs per liter. Treated human sewage contains 20-60 mg of BODs per liter, and raw human sewage contains 300-400 mg of BODs per liter (Burkholder et al. 2007).

Laws regarding the treatment and disposal of animal waste differ between countries. In Norway, disinfection of sludge has been mandatory since 1995 (Odegaard, Paulsrud, and Karlsson 2009). In the EU, laws regarding disinfection are being proposed. Land deposition of animal waste is the most common method of disposal in the US (EPA 2004). In Europe, 35-45% of animal waste is deposited on land, but land deposition is being phased out. In Norway, the use of disinfected biosoils is prohibited in areas where vegetables, berries, potatoes, or fruits are produced. In the EU, use in forests is prohibited (Odegaard, Paulsrud, and Karlsson 2009). For treating pathogens in animal waste, there are a number of effective methods, but in the case of unexpected waste spills the high concentrations of nutrients, heavy metals, and pathogens present in waste can contaminate surface and ground waters.

The properties of manure differ between animals, and the differences in manure types can affect the type of waste treatment or waste storage method used. Manure from swine, dairy cows, and layer chickens are typically handled as a liquid, which requires liquid storage methods. Manure from broiler chickens, turkeys, and beef cows is typically handled as a solid. In general, liquid wastes are more suitable for anaerobic digestion and solid wastes are more suitable for dry composting. Poultry manure has seen the largest increases for any category of animal waste: From 1982 to 2001, manure from poultry increased by more than 80%. Poultry manure is higher in phosphorus than other animal wastes, which has implications for treatment and deposition of the waste, because certain nitrogen-to-phosphorus ratios are desirable for plant growth (EPA 2004). Manure also changes characteristics after leaving the animal. The most prominent of these changes is the loss of nitrogen as ammonia into the air. By the time the manure is applied to soil, nitrogen losses may be 90%. These changes adversely affect the fertilizer value of manure. Loss of nitrogen to the air may also be a pollution concern if it gets redeposited in other watersheds, and airborne ammonia will cause bad odors, which becomes an annoyance to people in the community. The method of land deposition affects how much nitrogen is lost to the air – the loss is maximized with sprayers and minimized with direct injections into the soil. Phosphorus does not get lost to the air as nitrogen does, so the method of land deposition affects the nitrogen-to-phosphorus ratio as well (EPA 2004).

Besides manure, effluents from dairy and meat processing factories can also be included in the category of animal waste. Large quantities of water are used in dairy and meat processing operations. This water is used to clean the outsides and insides of carcasses, and for cleaning equipment and facilities before and after killing. There are three types of wastewater from dairy processing: sanitary wastewater, process wastewater, and cleaning wastewater. Sanitary wastewater is disposed of directly into municipal sewer systems. Processing wastewater is mainly used for cooling and heating. This water does not normally have pollutants and can also be disposed of in sewer drains. Cleaning water – used for cleaning facilities and equipment – regularly comes in contact with, and is mixed with, milk products. This type of wastewater is high in organics, is mixed with cleaning compounds, and is high in sodium (Liu and Haynes 2011). In meat processing plants, wastewater comes from stockyard washdowns and stock watering, rendering operations, fecal removal from intestines, and hair scalding. Effluents from these plants contains high organic compounds, high fat concentrations, high levels of N, P and Na, and it fluctuates in pH and temperature This wastewater effluent, like manure, also contains high levels of nitrogen and phosphorus. For farmers, the effluent can be a free source of nutrients, but use of the effluent on farmland carries with it environmental dangers such as dissolved salts causing soil salinity and groundwater contamination. These effluents can be treated in a two-stage process – first with screens to remove large debris, then with anaerobic or aerobic digestion.

2.2 Food Waste

Food waste in the US has increased approximately 50% since 1974, reaching 1400 kcal per person per day or 150 trillion kcal per year. Food waste is the biggest category of waste in the US (Zhang et al. 2006). An average US family of 2.63 persons generates 100 kilograms of wet food waste in slightly over a year (Diggelman and Ham 2003). Including restaurants and institutions as well as households, US food waste amounts to 43.6 million tons per year (Zhang et al. 2006). In the US, over one quarter of total freshwater supply is used on wasted food, and wasted food accounted for 4% of total US oil consumption in 2003 (Hall, Dore, and Chow 2009). Food waste is 30% solids and 70% water. The solid portion of the waste is 95% decomposable and 5% ash. The energy contained in wet food waste is 4650 kilojoules per kilogram (Diggelman and Ham 2003). This waste can be converted into energy in a variety of ways, including incineration with energy recovery, pyrolysis and gasification, and anaerobic digestion.

Food waste can be divided into three main categories: post-harvest losses of perishable and non-perishable crops, waste from food processing and retail, and post-consumer waste. Different countries, and different types of economies, have different sources of food waste. In developed economies, post-consumer food waste accounts for the most losses, and "as much as half of all food grown is lost or wasted before and after it reaches the consumer" (Lundqvist et al. 2008). In industrialized countries, post-harvest losses of non-perishable food crops are not significant. In other countries, 15% of grain may be lost in the post-harvest system. The post-harvest losses of perishable food crops are greater than non-perishable losses in both developing and developed countries. In the US, these losses are estimated to be (depending on the commodity) between two to 23 percent. The overall average post-harvest food loss for perishable crops in the US is 12 percent. In the UK, these losses are estimated to be 9 percent. In developing countries, current methods of fruit harvesting (leading to bruised fruit) may lead to greater losses if supply chains get longer; with current short supply chains, the bruised fruit always finds a consumer (Parfitt, Barthel, and Macnaugton 2010). For example, 30 percent of fresh fruit and vegetable production in India is lost through lack of a temperature-controlled supply chain (Mittal 2007).

In the UK, food and drink waste is estimated to be approximately 14 megatons. The largest source of waste in the UK is from households. For the food and drink manufacturing and processing sector, food waste is approximately 2.6 megatons. At the retail and distribution stage, the number is 366 kilotons per year (Parfitt, Barthel, and Macnaugton 2010).

In the UK, Post-consumer food waste amounts to 8.3 megatons of food and drink each year. This waste has a retail value of 12.2 billion pounds (in 2008 prices). This waste is equivalent to over 20 megatons of carbon dioxide emissions. The total percentage of food wasted in the UK is estimated at 25 percent of food purchased. In the US, Kantor et al. estimated that 25% of food is wasted. The EPA estimated that food waste was 12.7% of municipal waste, or 31.79 megatons. In Australia, food waste is estimated at 15% of the 20 megatons of municipal waste that goes to landfills each year. Dutch consumers throw away 8-11% of food purchased. A study by REFORSK found that food waste from households in 6 municipalities in Sweden in 1998 was 40.4% of total household waste (by weight). A study by Olsson and Retzner (1998) found that food waste was 14.8% of total household waste. This amounted to 177,682 dry tons/year (Finnveden et al. 2000). In a South Korean study of municipal waste, food waste was found to be 26-27% of household waste (Parfitt, Barthel, and Macnaugton 2010).

Several trends affect worldwide food waste. Urbanization in developing countries requires that new food supply chains be developed. The structure of these supply chains will affect future food waste. In BRIC countries (Brazil, Russia, India and China), diets are also changing to include less starchy food and more meat, dairy, and fish. These foods are more perishable. Also, transitioning to longer food supply chains may bring more waste; in the UK, "contractual penalties, product take-back clauses and poor demand forecasting had a combined influence that drove 10 per cent over-production and high levels of wastage in the UK FSC [food supply chain]" (Parfitt, Barthel, and Macnaugton 2010).

It cannot be assumed that all food waste goes into municipal solid waste. Household sink food disposers or "garbage disposals" divert the solid waste into wastewater systems. Of the 100 kilograms of wet food waste that an average US family generates in slightly over a year, 75% can be disposed of in a food waste disposer. 1031 kilograms of water is needed to flush 100 kilograms of food waste through the food waste disposer, and this is added to the food waste total (Diggelman and Ham 2003).

Waste cooking oil is a source of food waste used to produce biodiesel Waste cooking oil is high in abundance (˜9 pounds/person/year) and low in cost (between $0.09 and $0.20/lb)—Kulkarni and Dalai (2006). Its properties, however, differ based on cooking process, source collection, and storage conditions. When compared to petroleum-based diesel, Kulkarni and Dalai (2006) argue that emissions of HC, CO, NO, SO_2 and CO_2 decrease but NOx increase observed. Further, although engine performance is the same, biodiesel consumption is slightly higher (while the volumetric energy content of biodiesel reported in the literature is 33.3 to 35.7 MJ/L, petroleum based diesel is 40.3 MJ/L), but that blend of biodiesel and petroleum-based diesel (75:25 and 50:50) maintains balance for fuel consumption and emissions

Another source of oil used to produce biodiesel is restaurant Waste Lipids (Canakci, 2007). Canakci analysis suggests that 1 lb of fats and oils can be converted to 1 lb of biodiesel (1:1 ratio), and that the samples of feedstocks analyzed had moisture levels, which varied from 0.01% to 18.06%, and Free Fatty Acids content that varied from 0.7% to 41.8%. Yet another source used to produce biodiesel is soapstock, a byproduct of edible oil refining, which some argue is less expensive than edible-grade refined oils—it costs $0.11/kg, 1/5 price of crude soybean oil

2.3 Crop residues

Crop residues are a major byproduct of agriculture. These residues are cellulosic plant material such as corn stalks, corncobs, wheat straw, and rice chaff. Corncobs make up 20% of corn crop residue by weight. It is estimated that US production could produce 40-50 million tons of corn cobs per year (Perry 2013). For the entire world, the UNEP states that “140 billion metric tons of biomass is generated every year from agriculture” (UNEP 2009). Biomass is defined as all agricultural wastes, manures, and forestry scraps. This biomass is equivalent to approximately 50 billion tons of oil and could provide energy to 1.6 billion people in developing countries. In these countries, agricultural biomass is mostly left to rot in fields or burned.

Currently, biomass contributes 9-13% of the world’s energy supply. There has recently been great interest in converting energy-crops into biofuels and other types of energy, but their efficacy is limited by space constraints, growing conditions, and competition with food crops. Crop residues may be a better option – they do not interfere with food production, they can be harvested on a large scale in different climates, and “they can be harvested sustainably without affecting soil quality” (Donaldson et al. 2011). Agricultural residues such as corn stover, crop straw, and sugar cane bagasse show great potential for the production of energy. Corn stover particularly is predicted to play an important role in bioethanol production (Kim and Dale 2003). According to the US National Renewable Energy Laboratory, it is possible to produce 288-447 liters of ethanol per dry ton of corn stover.

A simple method of using crop residues for energy is to burn them in a a boiler to create steam and electric power. However, the efficiency of this process is limited by the high water content of the biomass. The most versatile method, according to Donaldson et al. (2011), is gasification and pyrolysis. The authors designate three different products generated from this process: high-pressure steam, electricity, and steam for the conversion of char into activated carbon. Pyrolysis involves subjecting the biomass to high temperatures in an oxygen-deficient environment. Activated carbon is in high demand for industrial applications, water purification, and air pollutant removal.

From the 5% of corn left on fields, 9.3 gigaliters of ethanol could be produced. Furthermore, the dry milling process of producing bioethanol produces dry grains that can be used as animal feed. These grains would replace a certain amount of corn as animal feed. If this corn were used for bioethanol instead of animal feed, another 5.3 gigaliters would be produced. This could replace .93% of world gasoline consumption. Allowing for 60% ground cover, the remaining available corn stover could produce 58.6 gigaliters of bioethanol. This could replace 3.8% of world gasoline consumption.

In the same way, bioethanol from barley waste and residue could replace 1.3% of global gasoline consumption. Oat waste and oat straw could produce 3.16 gigaliters of bioethanol, or 0.2% of global gasoline consumption. Energy produced from rice waste would amount to 14.3% global gasoline consumption, from wheat and wheat straw 7.5%, from sorghum 0.3%, and from sugar cane waste and sugar cane bagasse 3.4%. Furthermore, lignin residues from the process of creating ethanol could be used for generating electricity and steam. This electricity would amount to 0.7% of total global electricity generation. The total potential replacement of global gasoline consumption from crop residues is 32%, and the potential replacement of global electricity production is 3.6% (Kim and Dale 2003).

The use of crop residues for energy production is not without potential environmental issues. Soil conservation is a major consideration. Conservation tillage practices state that at least 30% of soil surface be covered with crop residues. In the US today, 90% of corn stover is left in the fields. Some authors argue that the best use of crop residues is to leave them on the fields, due to the nutrients and erosion-management benefits they provide to the environment. It is argued that the crop residues will nourish the soil and increase crop yields in the following season (Lal 2004). Others estimate that 30% of crop residue on the field can be harvested without affecting soil quality (Donaldson et al. 2011).

2.4 Sewage waste

Sewage sludge is a mixture of biosolids and liquids from wastewater treatment plants. Sludge is an unwanted byproduct of wastewater treatment, and it presents the challenges of disposal and disinfection. However, sewage sludge also has the potential to be used for beneficial means, such as fertilizer or energy production. Its high organic, nitrogen, and phosphorus content makes it suitable for this purpose (ECJRC 2001). In the US, over half of biosolids that are a product of municipal wastewater treatment go toward fertilizing or conditioning land, and the remaining portion is either incinerated or landfilled. Dumping sewage sludge into the ocean is no longer allowed (EPA 2004). In the EU, policies exist to enhance the use of sewage sludge in agriculture. However, there are regulations concerning acceptable levels of certain compounds in sewage sludge, depending on the country. The amount of sewage sludge produced is relatively constant – it is estimated to be about 50g of dry matter per person per day (Rulkins 2007). The EU member states produce 8 million metric tons of sewage sludge each year (ECJRC 2001). The presence of sewage sludge, and the treatment of it, will continue to pose environmental challenges into the future.

The use of sewage sludge as fertilizer is potentially very beneficial, as the sludge has the ability to put nutrients into the soil and replace some amount of chemical fertilizers. However, sludge carries a wide variety of pathogens that are infectious to humans as well as various species of animals and plants (ECJRC 2001). The treatment and disinfection of sewage sludge is therefore an important process, but it can be costly. Sewage sludge treatment accounts for over half of total wastewater treatment costs (Rulkins 2007).

There has been a large amount of interest over the past 2 decades in converting sewage wastewater effluent to energy in various forms. The main two options for producing energy from sewage sludge are incineration with energy recovery and anaerobic digestion. Heat from the incineration process can be recovered and used as energy. In anaerobic digestion, methane is produced which can be used to power generators to produce electricity.

2.5 *Other Organic Industrial, Municipal, and Residential Waste*

Municipal solid waste is composed of organic and non-organic portions. Paper and organic waste are the major components of the municipal solid waste stream by weight. Plastics make up 10% of the waste weight but 40% of the volume. The amount and composition of solid waste differs between developed and developing countries. According to one estimate, developed countries produce an average of 1.5 kilograms of municipal solid waste per person per day (Rubio-Romero et al. 2012). In developed Asian countries, solid waste generation is estimated at 0.4-0.6 kg per person per day. In developing Asian countries, the number is 0.7-0.8 kilograms per person per day. Also, recyclable waste is more common in developed countries, and organic waste is more common in developing countries (Othman et al. 2012). Global MSW is estimated at 1,100 million tons per year (Rubio-Romero et al. 2012).

The methane emissions from municipal solid waste in landfills are a major concern if they are not trapped and used as fuel. Global average emissions of methane into the atmosphere from MSW alone are estimated at 34 million tons per year (Rubio-Romero et al.

2012). Landfill emissions are estimated to be in the range of 6-20% of total world methane emissions (anthropogenic or otherwise). In the EU, landfill methane is required to be either used as fuel or flared. Flaring refers to the burning of methane, which converts it into carbon dioxide – a less potent greenhouse gas. Besides the greenhouse gas potential of methane from landfills, improper disposal of waste can also have far-reaching environmental and health effects such as ground and surface water contamination, soil and air pollution, spreading of diseases, and bad odors (Othman et al. 2012).

3 Waste Management Technologies

A waste hierarchy is often used in waste policy. This hierarchy is: 1. Waste reduction, 2. Reuse, 3. Recycle, 4. Incinerate with heat recovery, 5. Landfill. This hierarchy can be seen in certain pieces of legislation, such as the German “Law on the Prevention and Disposal of Waste” (1986), followed by the “Closed Loop Economy Law” (1994); the US “Pollution Prevention Act” (1990); and in Denmark, the “Government Action Plan on Waste and Recycling” (1993), which set targets for recycling, incineration, and landfill. After waste reduction, the priorities of waste treatment are often contested (Finnveden et al. 2000)

3.1 Anaerobic digestion

There is a question of where to place anaerobic digestion and composting in this hierarchy. Anaerobic digestion is an effective and widely used treatment strategy, which utilizes microorganisms to digest waste without the presence of oxygen. Four types of microorganisms are used in anaerobic digestion: hydrolytic, fermentative, acetogenic, and methanogenic (Braber 1995). Anaerobic digestion produces methane (CH4) and carbon dioxide (CO2), as well as trace amounts of other gases. Anaerobic digestion produces 60-70 percent methane and 30-40 percent carbon dioxide (USDA 2009). Anaerobic digestion is also effective in disinfecting waste, which is a mandatory process in countries such as Norway, and could become mandatory in the EU. The biogas products of anaerobic digestion can be used to generate heat or electricity, and the resulting digestate can be used as fertilizer, which can be applied in agriculture. Mata-Alvarez et al. (2000) analysis suggests that while emissions of VOC from composting are 747.4 g/ton biowaste, anaerobic digestion is only 100.6 g/ton biowaste. Furthermore, they document a reduction of emissions with anaerobic digestion of CO_2 emissions of 25 to 67% than composting.

Methods of anaerobic digestion include covered lagoons, complete mix digesters, and plug-flow digesters (EPA 2004). Covered lagoons are a wet digestion system that involves digesting the waste underneath a covering which traps biogas emissions. Complete mix digesters allow inflows of new material to be mixed with the partially digested material and plug-flow digesters do not allow new material to be mixed with the biomass undergoing digestion. After digestion, the digestate is usually refined for use on farmland or compost, and some portion of the liquid effluent will go back into the digestion process as inoculum (Braber 1995).

3.1.1 Animal waste

With anaerobic digestion, farmers of livestock can obtain energy from biogas, fertilizer and, in countries such as Brazil (Kunz, Miele, and Steinmetz 2008), even carbon credits. Anaerobic digestion is also used to treat effluent wastewater from dairy and meat processing factories. Anaerobic lagoons are common where land is available for their construction and where the climate is warm. They are not common in urban areas. Suspended biomass reactors include anaerobic contact reactors and suspended biomass reactors. Anaerobic contact reactors are considered dated technology but are still used in older and smaller factories throughout the world (Liu and Haynes 2011). Anaerobic sequencing batch reactors have shown high efficiency in treating wastewater. Immobilized cell reactors—i.e., cells that are entrapped within or associated with an insoluble matrix—have been successfully applied in full-scale operations for almost 2 decades (Liu and Haynes 2011).

Around the world, the most common disposal strategy for animal manure is land deposition. In Brazil, the most common treatment strategy is anaerobic digestion (Kunz, Miele, and Steinmetz 2008).

3.1.2 Food waste

For the most part, food waste in the US goes to landfills, but energy-from-waste is becoming more economically viable. Due to the high moisture content of food waste, anaerobic digestion is preferred to combustion or gasification (Zhang et al. 2006). Zhang et al. (2006) concluded that "...food waste contained the required nutrients for anaerobic microorganisms." and that it was "a highly desirable feedstock for anaerobic digestion."

In one study by Finnveden et al. (2000), electricity consumed in the anaerobic digestion process was 31 megajoules per ton of organic household waste (dry weight). Dry weight is 30% of wet weight. The plant's heat consumption in the study was 495 megajoules per ton organic household waste (dry weight). This heat was generated from the methane gas produced. The digestion residue contained 7.6 kg/ton nitrogen and 1.1 kg/ton phosphorus (dry weight). Describing the energy needed for spreading the produced residue from anaerobic digestion and composting, the authors say "The energy required for spreading is approximately 20 MJ diesel/ton digestion residue and 15 MJ/ton compost" (Finnveden et al. 2000). From 1 ton food waste, anaerobic digestion produces 858kg residue (14.2% weight loss, wet weight). Composting produces 500kg (50% weight loss, wet weight). The avoided energy costs of producing and spreading artificial fertilizers must also be taken into account in the life-cycle analysis. The energy produced from methane from food waste is 3743 MJ/ton food waste, and 495 MJ of that is used for the plant itself (Finnveden et al. 2000).

It is also possible to co-digest food waste and yard waste, which possibly gives greater biogas yields. Co-digestion of other wastes (food waste with dairy manure) has been shown to increase total methane production. 30.9 million metric tons/year of yard waste (grass, leaves, wood chips) is available for co-digestion. Brown and Lee 2013 attempted to determine the optimal combination of food waste and yard waste for maximum yields of methane. The study showed that, at F/E ratio of 2, increased methane production was seen when food waste was 10% of the substrate. At F/E ratio of 1, increased methane production was seen when food waste was 20% of the substrate.

Oil waste, a type of food waste, can be used to produce biodiesel, and several alternative technologies are discussed in the literature. One of those technologies is Supercritical Methanol—a catalyst free technology of biodiesel production under high temperature and pressure. Different from existing technologies, it is insensitive to water and Free Fatty Acid content. When using this technology, the feedstocks can account for 80% of total costs of production, biodiesel produced using Supercritical Methanol is similar to commercial diesel fuel, and the reaction time is relatively short (Demirbas 2009). Lee et al. (2011) evaluated 3 biodiesel processes:

1. Alkali-catalyzed, virgin oil (Alakai-FVO);
2. Two-step process, WCO (Alkali-WVO); and
3. Supercritical Methanol process, WCO (SC-WVO).

Their analysis suggests that energy consumption of alkali-FVO of 2349 kW, alkali-WVO of 5258 kW, and SC-WVO of 3927 kW. Those authors conclude that energy consumption of SC comparable to Alkali-FVO, and that the most significant variable affecting production cost is feedstock (64-84%). Using co-solvent in Supercritical Methanol, Cao et al. (2004) conclude that because Supercritical Methanol requires temperatures of 350-400^0 C and pressures of 45-65 MPa, this process is one with high production costs and energy consumption. The analysis presented by those authors suggests that using propane as a co-solvent reduces severity of necessary conditions: At 280^0 C and 12.8 MPa, they obtained methanol/oil ratio of 24 and 98% yield within 10 minutes. Thus, the authors conclude that the Supercritical Methanol with co-solvent is superior to Supercritical Methanol by itself, and that when using co-solvent lower temperatures and pressures is required, and that less energy required for process.

When using Waste Cooking Oil, Kulkarni and Dalai (2006) argue that Chemical Transesterification Processes: (i) Alkali-Catalyzed Transesterification has fast reaction rate but is inefficient when Free Fatty Acid content is high; and (ii) Acid-Catalyzed Transesterification has a slow reaction rate but works better if Free Fatty Acid content is high. On the other hand, a Two-Step Acid and Alkali Transesterification, whereby

1. Stage 1: pretreatment process (acid-catalyzed)
2. Stage 2: transesterification (alkali-catalyzed),

has problems with pretreatment of feedstock, recovery of glycerol, removal of catalyst, and energy intensity. Other Transesterification Processes surveyed, included Enzyme-Catalyzed Transesterification, where the authors argued that this process has no generation of by-products, easy product recovery, mild reaction condition, and is insensitive to FFA and moisture content. Another process discussed and analyzed by the authors was a Catalyst-Free Technology, i.e., the Supercritical methanol process—an emerging technology that results in higher yield of biodiesel even with high Free Fatty Acid and moisture content. However, scaling up and commercialization has yet to be proven. Testing of Biodiesel Produced from Waste Cooking Oil, the presented analysis suggested estimated costs for biodiesel from:

1. Soybean oil - $0.418/L
2. Yellow grease - $0.317/L; and
3. Brown grease - $0.241/L.

The author concluded that chemical processes are not recommended due to high costs. But that enzyme transesterification may be a good option that should be developed. The authors also concluded that Supercritical methanol method has potential but scaling up is required. Canakci (2007), while focusing on Restaurant Waste Lipids concludes that because moisture and Free Fatty Acid content vary widely, the Transesterification processes needs to account for this, and that Two-step process (alkali and acid transesterification) is recommended.

Diaz-Felix et al. (2009) looked at Yellow Grease, and seek to quantify output based upon Free Fatty Acid content of feedstock and methanol use during reaction. They also advocate for a two-step process, with pre-treatment followed by transesterification. Their analysis suggests that while using molar ratio of 18:1 and 95% conversion, the Fatty Free Acid content were reduced to 0.4%. The authors further argues that this process can be further optimized as only 85% conversion is necessary for reduction in Fatty Free Acid content to 2%, and that the alkaline-catalyzed transesterification successful at 2% content. Hass (2005) also argued for the two-step process, while focusing on Soapstock–a byproduct of edible oil refining. Hass argued that the Saponification process

1. Step 1: alkali-catalyzed hydrolysis increases FFA; and
2. Step 2: acid-catalyzed transesterification produces FAME,

is efficient and that it met ASTM specifications for biodiesel and the product was comparable to biodiesel from soy oil. It costs $0.41/L to produce biodiesel from soapstock, which is 25% less than production from soy oil.

3.1.3 Crop residues

In Europe, biogas plants are mainly fed with animal waste effluents and dedicated energy crops. As the number of biogas plants has increased, the amount of dedicated energy crops has also increased. There are concerns about world food supply in this system. As an alternative, crop residues can be used, and there will be no food supply concerns (setting aside the argument that taking crop residues off the fields can reduce subsequent crop yields). If the residues are harvested sustainably, leaving enough on the fields for effective nutrient deposition and erosion control, then this will not be a concern.

Combustion-based plants are rare in Italy due to the fact that the production of thermal energy is not incentivized, but the government incentivizes electricity production. Thus, the thermal energy is converted into electricity by heating water to create steam to power turbines. The efficiency of this process is less than using methane-powered engines which would be used at biogas plants. Combustion plants also require flue gas purification, but anaerobic digestion biogas plants do not. And the digestate from biogas plants can be used as fertilizer. Using crop by-products instead of energy crops also reduces greenhouse gas emissions (Menardo and Balsari 2012).

As the result of a study by Menardo and Balsari (2012) which compared methane yields of different types of crop residues, maize stalks produced the lowest yields. Rice chaff, wheat straw, kiwi, and onion were not significantly different in yields. Dairy products produced the most biogas and methane yields due to high protein and fat content – however, dairy products are not available in large amounts, and would not be applicable on a large scale. Dry bread biogas yields were not significantly different from rice chaff, maize stalks and wheat straw but had a high methane percentage compared to other gases. The lignocellulosic compounds in corn stalks and wheat straw negatively affected methane yields, but the huge amounts of these resources that is available for use makes them attractive energy sources nonetheless. Rice chaff also has high availability in Italy and it is also an attractive potential source of biogas. Mechanical shredding can be used as pre-treatment for these types of crops to prepare them better for digestion (Menardo and Balsari 2012).

3.1.4 Sewage waste and municipal solid waste

In a typical situation, sewage waste is mechanically and biologically treated with sedimentation followed by anaerobic digestion. This process involves 3 steps: a hydrolysis step, an acidification step, and a third step in which biogas is produced. Anaerobic digestion is used to stabilize the sewage and convert it into biogas. Menger-Krug et al. (2012) state that, "Currently, anaerobic digestion of sewage sludge is mainly applied at large and medium-sized wastewater treatment plants. However, also, a growing interest is observed in the application of anaerobic treatment in small-sized plants." Pretreating the sewage in a variety of ways can also increase the amount of biogas produced.

The portions of municipal solid waste that are suitable for anaerobic digestion are paper, green waste or yard waste and food waste (Hanandeh and El-Zein 2010). The organic fraction of municipal solid waste is approximately 50% (Braber 1995). The typical method of disposal for municipal solid waste is incineration or landfill. Anaerobic digestion has been used for over 100 years to treat sewage sludge, but experience with digesting the organic portion of municipal solid waste is more recent. Typically, anaerobic digestion is most appropriate when the waste is mostly food waste, and aerobic composting is most appropriate when it is mostly yard waste. Anaerobic digestion provides a good solution for treating the wet fraction of municipal solid waste which is too high in moisture for incineration. This method also requires less land than aerobic composting. Furthermore, because anaerobic digestion can be applied to many different types of waste, it is possible to co-digest the organic portion of municipal solid waste with other wastes such as agricultural manure (Braber 1995). The efficiency of the digestion process is even increased in some cases when wastes are co-digested (Brown and Lee 2013).

In order for municipal solid waste to undergo anaerobic digestion, it must be pre-treated, which includes sorting and separation of differently sized particles. In wet systems, sorting and separation can occur simultaneously. For dry digestion systems, rotating drums are typically used. For wet digestion systems, pulpers are used (Braber 1995).

3.2 Waste Incineration

3.2.1 Sewage waste

Incineration with energy recovery is a process that can be applied to mechanically-dewatered or dried sludge. The sludge before dewatering is 98% water. The dewatering techniques include drying beds, belt filter presses, plate and frame presses, and centrifuges. Pretreatment strategies can also be used before dewatering, which involves mixing the sewage sludge with lime, ferric chloride, or other chemicals (EPA 2004). The dewatering process adds to the cost of incineration.

There are potential environmental problems with incineration, involving emissions of pollutants and quality of ash produced. There are systems (scrubbers) to eliminate the harmful pollutants from the emissions, but they are costly. This is one of the main reasons why incineration is an expensive option compared to other treatment methods. The heavy metals in the ash are not an environmental problem because, due to the high heat treatment, they are immobilized and not prone to leaching. Currently, incineration of sludge is mainly used for generating heat (steam) or electricity. Incineration is used worldwide and on a large scale. The performance largely depends on the water content and the performance of the drying process. To reduce costs, the dried sludge can be co-incinerated in an existing coal-fired power plant. The combustion and the gas treatment systems will be already in place, so initial costs will be lowered. This option is already applied in practice (Rulkins 2007).

The ash from the incineration process can be used in the production of various products such as bricks or portland cement. This process is used today in Japan. The energy efficiency of the production process is not very high; it has high costs, and the current practical applications of this technology are limited (Rulkins 2007)

3.2.2 Food waste

Incineration is expanding in Sweden: “The amount of waste incinerated has more than doubled since 1980 and the energy production has increased almost fourfold during the same period” (Finnveden et al. 2000). In 1995, 1.8 megatons of waste were incinerated, and the energy produced from this waste was 5 terawatt-hours (Finnveden et al. 2000). It is possible that incineration of food waste offers no net energy benefit due to the high moisture content in food waste, which requires energy for drying or dewatering. However, some authors argue that the moisture can be evaporated off efficiently and that net energy benefits can be attained (Bernstad and la Cour Jansen 2012).

3.3 Municipal solid waste

Incineration is an option for managing municipal solid waste. Incineration is effective in reducing the volume of the waste to be disposed and sterilizing it. Volume reductions of the waste are up to 90%. However, strict emissions regulations have made this option more costly. Nonetheless, some countries are attempting to reduce the amount of organic content of post-recycled waste destined for landfill and promoting incineration as an alternative. If the infrastructure for incineration and emissions purification are already in place, such as with co-incineration inside coal-fired power plants, then this option becomes more economical.

Many countries have deemed “bottom ash” (the ash collected during incineration – contrasted with the finer and more leaching-prone “fly ash”) suitable for disposal in landfills. But it is also used in construction materials. Germany, Denmark, and Sweden use 60-90% of bottom ash in road construction or in asphalt/concrete. The fly ash, by contrast, is considered hazardous and needs to be disposed separately. However, in the US, the fly ash and bottom ash are combined and disposed of together (Sakai et al. 1996)

3.4 Composting

Composting can be used to treat a variety of organic wastes, including animal manure, sewage waste, the organic portion of municipal solid waste, food waste and yard waste. Composting is commonly used to treat these wastes in the US. Composting reduces the volume of the waste as well as disinfecting and stabilizing it. Reduced volume and weight leads to easier and more economical transportation of the waste product. After the composting process, the compost can be used to fertilize land or be placed in a landfill. However, composting does not result in usable amounts of methane. The process is aerobic and mostly results in the production of carbon dioxide. From 1990 to 2010, the amount of waste composted in the US rose 392 percent. This increase is mainly attributable to population growth and legislation in the 1990s to reduce the amount of yard trimming disposed in landfills (EPA 2010). Sakai et al. (1996) stated that biological treatment methods were reemerging as commercially viable, which had led toward composting garden, kitchen, and commercial food wastes, and away from mixed solid waste processing.

Large amounts of composting can be done through aerated static pile, aerated windrow, or in-vessel composting. Aerated static pile composting simply involves one large pile of waste mixed together with layers of bulking agents such as wood chips, which allow air to reach the bottom of the compost pile. Aerated static pile composting works well for yard waste, food waste and paper, but not for animal waste. Aerated windrow composting is different from static pile composting in that the waste is placed in rows which get turned over periodically. In-vessel composting involves depositing the waste in large drums, silos, or trenches and controlling the moisture and aeration of the waste mixture. This method of composting works well for all types of organic waste, although, like all aerobic composting, it does not have the potential to produce useable amounts of methane from the waste.

Composting (especially in-vessel composting) solves many problems for farmers who need to manage large amounts of solid or liquid manure. Composting transforms liquid manure into solid, which reduces the volume and weight of the manure and lessens its transportation cost (Kunz, Miele, and Steinmetz 2008). Transportation cost is one of the main considerations in animal waste management because the waste must be transported off the farm site due to the inability of the on-site farmland to assimilate the nutrients present in the waste. Composting is also effective for disinfecting waste. The common methods of waste disinfection are as follows (Odegaard, Paulsrud, and Karlsson 2009):

- Lime treatment
- Dry composting
- Wet composting
- Wet composting + anaerobic digestion
- Pre-pasteurization + anaerobic digestion
- Anaerobic digestion + thermal drying

Composting for disinfection purposes can be performed in either a wet or dry state, or combined with anaerobic digestion.

In the composting of food waste, there are commonly 3 stages: mechanical pretreatment, composting, and manufacturing of soil products. The soil products produced can be used for fertilizer. The composting process on a large scale has implicit costs associated with it. One study found that electricity consumed in the process is 54.4 megajoules per dry ton of food waste. Also, consumption of diesel in the process is 555.5 megajoules per dry ton of food waste. The compost residue contained 8.3 kilograms of nitrogen per dry ton and 2.0 kilograms of phosphorus per dry ton.(Finnveden et al. 2000)

3.5 Landfilling

Landfills are the third-largest source of human-related methane emissions in the US. The EPA estimates landfill methane emissions to be 16.4% of total US methane emissions in 2010 (EPA 2012). This figure is contrasted with the percentage of methane emissions from wastewater treatment (2.5%) and composting (<1%). Clearly, landfills present a huge environmental issue: how best to deal with these emissions of methane, which is a greenhouse gas with 25 times the potency of carbon dioxide.

There are about 1,900 landfills operating in the US today. Over the past 20 years, the number of landfills has decreased and the average landfill size has increased. The process of decomposition in landfills is initially an aerobic one, but then turns anaerobic when oxygen is depleted. Eventually, this process leads to the production of methane and carbon dioxide in approximately equal parts. From 1990 to 2010, net methane emissions from landfills decreased 27%, even though municipal solid waste deposition in landfills increased 23%. This change is due to several factors. Even though the amount of waste increased, the amount of decomposable waste decreased by 21%. Methane-recovery in landfills also increased. From 2009 to 2010, there was an increase in methane recovery of 5%. In 2010, 54 new landfill gas-to-energy projects began. It is estimated, in part because of the rising US population, that municipal solid waste will increase in the coming years – but the percentage landfilled may decrease if more waste is recycled and composted (EPA 2012).

Food waste in the US, for the most part, goes to landfills, but energy-from-waste is becoming more economically viable (Zhang et al. 2006). Landfilling is also one of the most widely used options in Sweden with 300 municipal landfill plants. In 1998, 4.8 million tons of waste were landfilled. Energy gained from landfills is used for 60% heat and 30% energy, and 10% is lost. There are also more efficient types of landfills known as biocells (Finnveden et al. 2000).

Municipal solid waste landfills receive approximately 69 percent of the solid waste generated in the US. The remaining waste is sent to industrial landfills. However, the municipal solid waste landfills accounted for 94% of the total methane emissions. The US has banned many polluting compounds, and banned disposal of certain items like car batteries in landfills.

4 Discussion and concluding remarks

Many facets of converting biodegradable waste to energy and the associated technologies are mature and used extensively in some regions but not in others. These technologies can potentially reduce greenhouse gases and help countries achieve their Kyoto Protocol goals. However, the realm of adoption of biodegradable technologies is one of divergent experiences. Within Europe, Germany and Denmark were the early adopters of the anaerobic technology in the early 1990s, with much of Europe following. The US, however, only began to experience growth starting in 2003. Interesting but Europe does not have a comparative advantage in either technology or feedstock availability, although work has detected large differences in policy and especially in use of the feed-in-tariffs overtime (Bangalore et al., 2012).

More generally, the literature has shown that adoption is impacted by lack of access to credit, and that these constraints may hamper the introduction of otherwise profitable technologies (Zilberman et al. 2012). Furthermore, financial incentives have a positive effect on adoption of energy conservation technologies but that the elasticity of adoption in response to financial incentives is low (Linn, 2008). These conclusions can be explained using the putty-clay nature of capital-intensive assets and the long-term commitments associated with adoption of capital goods, where response to financial incentives that results in the adoption of new technologies is associated with the response to new entrants. Further, research has concluded that capital investments are barriers to adoption and use of agricultural anaerobic digestion among farmers (e.g., MacDonald et al., 2009; Bywater, 2011).

Experiences in Europe as well as to the west of the Atlantic indicate policy stability in the form of financial support is the most important factor in promoting the adoption of technologies converting biodegradable waste to energy. As more countries acknowledge the multitude of benefits from these technologies, adoption will likely increase. In future work we plan on further understanding the importance of policy and further explain differences in patterns of adoption of these technologies among countries and regions.

References

1. Bangalore, M., G. Hochman, and D. Zilberman (2012). “Differences in the Adoption of Agricultural Anaerobic Digestion in Europe and the United States”
2. Bernstad, A. and J. la Cour Jansen (2012). ”Review of comparative LCAs of food waste management systems: Current status and potential improvements.” Waste management.
3. Boon, A. (2002). ”Wastewater sludge- rubbish or resource?” Water & Sewerage Journal(3): 33-36.
4. Braber, K. (1995). ”Anaerobic digestion of municipal solid waste: a modern waste disposal option on the verge of breakthrough.” biomass and bioenergy 9(1): 365-376.
5. Brown, D. and Y. Li (2012). ”Solid State Anaerobic Co-Digestion of Yard Waste and Food Waste for Biogas Production.” Bioresource Technology.
6. Bywater, A (2011). A Review of Anaerobic Digestion Plants on UK Farms – Barriers. The Royal Agriculture Society of England.
7. Canakci, M. (2007). ”The potential of restaurant waste lipids as biodiesel feedstocks.” Bioresource Technology 98(1): 183-190.
8. Cao, W., H. Han, et al. (2005). ”Preparation of biodiesel from soybean oil using supercritical methanol and co-solvent.” Fuel 84(4): 347-351.
9. Demirbas, A. (2009). ”Biodiesel from waste cooking oil via base-catalytic and supercritical methanol transesterification.” Energy Conversion and Management 50(4): 923-927.
10. Diaz-Felix, W., M. R. Riley, et al. (2009). ”Pretreatment of yellow grease for efficient production of fatty acid methyl esters.” biomass and bioenergy 33(4): 558-563.
11. Diggelman, C. and R. K. Ham (2003). ”Household food waste to wastewater or to solid waste? That is the question.” Waste management & research 21(6): 501-514.
12. Donaldson, A. A., P. Kadakia, et al. (2011). ”Production of Energy and Activated Carbon from Agri-Residue: Sunflower Seed Example.” Applied Biochemistry and Biotechnology: 1-9.
13. Finnveden, G. r., J. Johansson, et al. (2005). ”Life cycle assessment of energy from solid waste part 1: general methodology and results.” Journal of Cleaner Production 13(3): 213-229.
14. Gebrezgabher, S. A., M. P. Meuwissen, et al. (2010). ”Economic analysis of anaerobic digestion: A case of Green power biogas plant in The Netherlands.” NJAS-Wageningen Journal of Life Sciences 57(2): 109-115.
15. Haas, M. J. (2005). ”Improving the economics of biodiesel production through the use of low value lipids as feedstocks: vegetable oil soapstock.” Fuel Processing Technology 86(10): 1087-1096.

16. Hall, K. D., J. Guo, et al. (2009). "The progressive increase of food waste in America and its environmental impact." PLoS One 4(11): e7940.
17. Institute for Environment and Sustainability Soil and Waste Unit (2001). Organic Contaminants in sewage sludge for agriculture use. Link: http://ec.europa.eu/environment/waste/sludge/pdf/organicsinsludge.pdf
18. JoAnn Burkholder, Bob Libra, Peter Weyer, Susan Heathcote, Dana Kolpin, Peter S. Thorne, and Michael Wichman. "Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality," Environ Health Perspect. 2007 February; 115(2): 308–312. Link: http://www.ncbi.nlm.nih.gov/pmc/articles/PMC1817674/
19. Kim, S. and B. E. Dale (2004). "Global potential bioethanol production from wasted crops and crop residues." biomass and bioenergy 26(4): 361-375.
20. Kulkarni, M. G. and A. K. Dalai (2006). "Waste cooking oil an economical source for biodiesel: A review." Industrial & engineering chemistry research 45(9): 2901-2913.
21. Lal, R. (2005). "World crop residues production and implications of its use as a biofuel." Environment International 31(4): 575-584.
22. Lee, S., D. Posarac, et al. (2011). "Process simulation and economic analysis of biodiesel production processes using fresh and waste vegetable oil and supercritical methanol." Chemical Engineering Research and Design 89(12): 2626-2642.
23. Linn, J. (2008). Energy prices and the adoption of energy-saving technology. The Economic Journal 118(553), 1986–2012
24. Liu, Y.-Y. and R. Haynes (2011). "Origin, Nature, and Treatment of Effluents From Dairy and Meat Processing Factories and the Effects of Their Irrigation on the Quality of Agricultural Soils." Critical Reviews in Environmental Science and Technology 41(17): 1531-1599.
25. Mata-Alvarez, J., S. Mace, et al. (2000). "Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives." Bioresource Technology 74(1): 3-16.
26. Menardo, S. and P. Balsari (2012). "An Analysis of the Energy Potential of Anaerobic Digestion of Agricultural By-Products and Organic Waste." BioEnergy Research: 1-9.
27. Menger-Krug, E., J. Niederste-Hollenberg, et al. (2012). "Integration of Microalgae Systems at Municipal Wastewater Treatment Plants: Implications for Energy and Emission Balances." Environmental Science & Technology 46(21): 11505-11514.
28. Othman, S. N., Z. Z. Noor, et al. (2012). "Review on life cycle assessment of integrated solid waste management in some Asian countries." Journal of Cleaner Production.
29. Parfitt, J., M. Barthel, et al. (2010). "Food waste within food supply chains: quantification and potential for change to 2050." Philosophical Transactions of the Royal Society B: Biological Sciences 365(1554): 3065-3081.
30. Rubio-Romero, J. C., R. Arjona-Jiménez, A. López-Arquillos (January 2013). "Profitability analysis of biogas recovery in municipal solid waste landfills." Journal of Cleaner Production
31. Rulkens, W. (2007). "Sewage Sludge as a Biomass Resource for the Production of Energy: Overview and Assessment of the Various Options." Energy & Fuels 22(1): 9-15.
32. Sakai, S., S. Sawell, et al. (1996). "World trends in municipal solid waste management." Waste management 16(5-6).
33. U.S. Environmental Protection Agency secretariat (2004). "Risk Assessment Evaluation for Concentrated Animal Feeding Operations," The U.S. Environmental Protection Agency. 2004.
34. U.S. Environmental Protection Agency (2004). Primer for municipal wastewater treatment systems. Link: http://www.epa.gov/npdes/pubs/primer.pdf
35. U.S. Environmental Protection Agency (2012). U.S. Greenhouse Gas Inventory Report: Chapter 8. Link: http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2012-Chapter-8-Waste.pdf
36. Zhang, R., H. M. El-Mashad, et al. (2007). "Characterization of food waste as feedstock for anaerobic digestion." Bioresource Technology 98(4): 929-935.
37. Zilberman, D., Zhao, J., and Heiman, A (2012). Adoption Versus Adaptation, with Emphasis on Climate Change. *Annual Review of Resource Economics* 4, 27–53