CZESTOCHOWA UNIVERSITY OF TECHNOLOGY

Environmental safety of biowaste in the circular economy

Editors
Ewa Neczaj
Anna Grosser

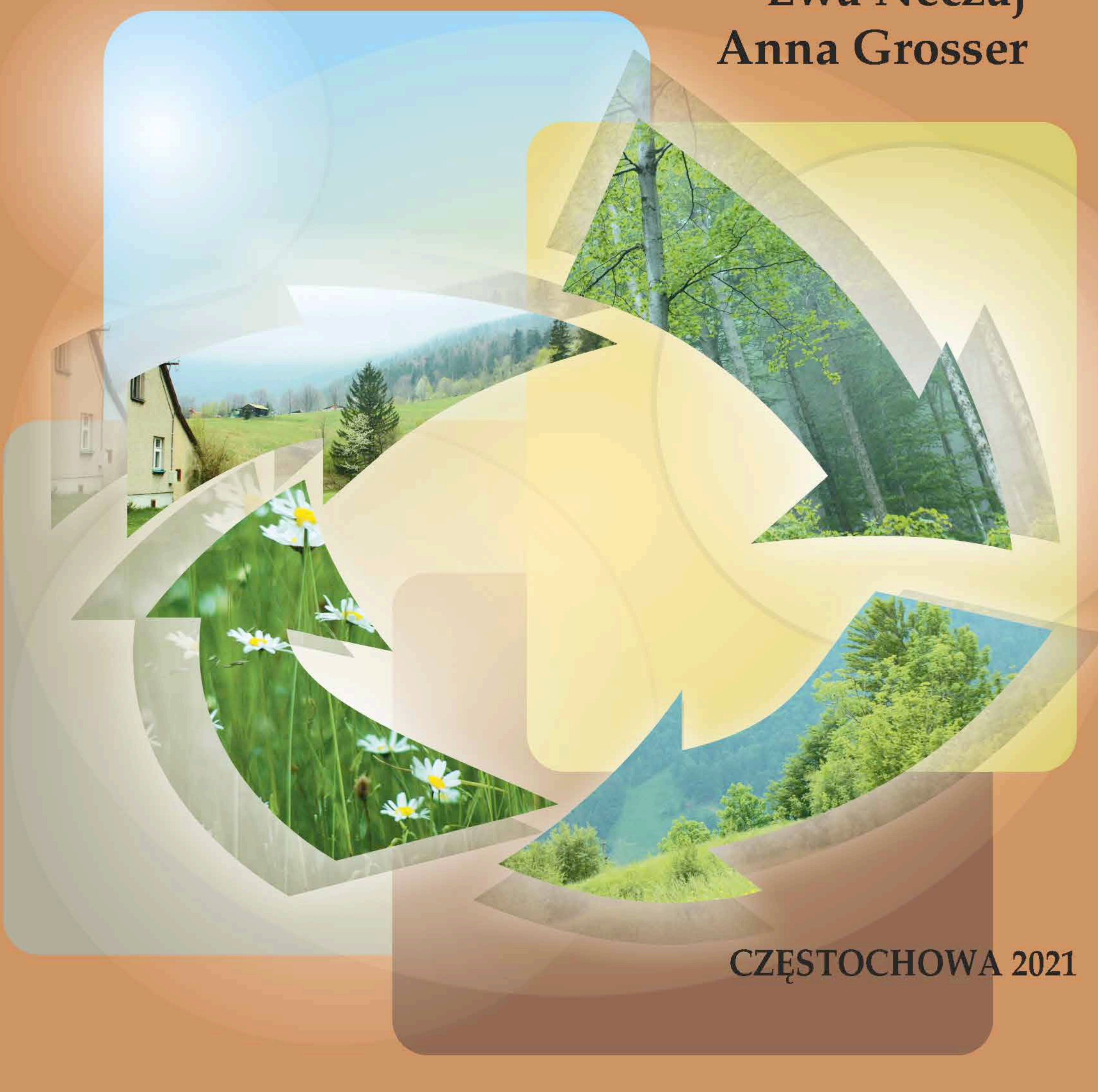

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Table of Contents

Part II
Methods used for treatment and final disposal od biowaste and sewage sludge

Part IV
Circular economy concept

Preface

If we want to stop climate change, it is necessary to intensify the implementation of the idea of a circular economy (CE). On March 2020, the European Commission adopted a new circular economy action plan based on implementation report of previous plan adopted in 2015. As highlighted in this paper to be climate neutral by 2050 "*EU needs to accelerate the transition towards a regenerative growth model that gives back to the planet more than it takes, advance towards keeping its resource consumption within planetary boundaries, and therefore strive to reduce its consumption footprint and double its circular material use rate in the coming decade*".

The book deals with a very important element of the circular economy related to sustainable management of biodegradable waste. According to World Bank by 2050, the human population is estimated to increase more than 9 billion; meanwhile, energy demand is expected to almost double, and the demand for water and food is expected to increase by about 60%.

It is also obvious that the amount of biodegradable waste will grow in the coming years, and it will become a global problem and need to change in the organic waste management system. If not properly handled, the large volume of biodegradable waste may deteriorate air, water, and soil quality, resulting in significant impacts to food, energy, and water supplies. However, waste also poses a great opportunity as feedstock for renewable energy generation and production of value-added products Thus, reuse and treatment of biodegradable waste play an essential role in the food-energy-water nexus.

The individual chapters of the book are aimed at the importance of biowaste and other biodegradable waste in a circular economy, taking into account environmental aspects (e.g. toxicity, organic carbon sequestration, LCA).

This book will serve as a useful resource for environmental engineers, biotechnologists, researchers, and students studying organic waste processes, as well as operators of a wastewater and organic waste treatment plants.

We strongly hope that readers enjoy reading this book and find it of very use.

Ewa Neczaj & Anna Grosser

Part I

Characteristic of biowaste/sewage sludge

Chapter 1

Identification of organic and inorganic contaminants presented in sewage sludge and biowaste

Monika GAŁWA-WIDERA

1.1. INTRODUCTION

The natural management of sewage sludge is conditioned by the provisions of the Waste Act and ordinances regarding sewage sludge with later legal acts, which contain the principles of handling this type of waste (Dz.U. Nr 62, poz. 628; Dz.U. Nr 137, poz. 984). Sewage sludge and biowaste are a group of waste that can be used, e.g. in nature or for the production of alternative fuels. The increasing amount of sewage sludge resulting from the intensive development of the economy and the urbanization process is one of the major environmental protection problems and may lead to an ecological and biological imbalance in nature.

The main factors responsible for the growing level of sewage sludge production in Poland are mainly the modernization and construction of new treatment plants and the extension of sewage networks (Bień, 2007; Bień, Neczaj, 2011). Observation shows that the amount of sewage sludge generated has been increasing steady in recent years.

Over the last few years, significant changes have been observed in the approach to problems related to the development of sludge management in sewage treatment plants. The principles adopted are the basis for actions:

- limiting the content of harmful and toxic substances in wastewater flowing into the treatment plant, which determines the composition of the resulting sludge,
- effective sludge treatment using appropriate methods from a technological, chemical, ecological and economic point of view to reduce their quantity and improve technological and operational properties,
- recovery of energy and selected products carried out in advanced processing and disposal processes,
- rational and safe sludge management, with preference for their natural use.

The type of wastewater flowing into the treatment plant determines the physico-chemical composition of the sludge. Some of these pollutants do not decompose in purification processes and accumulate in sediments. The level of these pollutants is generally not reduced in sludge treatment processes either (Barbusiński, 2016).

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These impurities are called potentially toxic and may include:

- heavy metals,
- polychlorinated biphenyls (PCBs),
- polycyclic aromatic hydrocarbons (PAHs),
- furans and dioxins (PCDFs),
- pesticides (Bernacka, 2000).

The possibility of wider research in the field of the abovementioned organic pollutants was associated with the development of instrumental analytical techniques, as well as better documentation of the toxic effects of these substances on animal organisms and humans (Hermanowicz et al., 1999). In addition, it should be emphasized that the identification of toxic organic micro-pollutants is difficult and quite expensive.

In Poland, legal acts have been introduced regarding the conditions to be met when introducing sewage into water or soil containing selected organic pollutants (Dz.U. Nr 134, poz. 1140). However, the regulation on the natural use of sewage sludge does not specify the number of organic pollutants that can be released into the environment, including soil, but only the level of heavy metals (Dz.U. Nr 134, poz. 1140).

The study is an auxiliary material for students of environmental engineering and biotechnology. Chemicals of organic and inorganic origin, depending on the environment and coexisting compounds, change over time.

In order to understand unit processes, it is necessary to know the structure, environmental impact, chemistry and properties of the individual elements discussed in the following chapter.

Organic and inorganic contamination of soil is a big problem in the field of soil remediation.

Knowledge of the chemical nature of a selected group of compounds is the basis for the development of a technology for cleaning polluted areas. The situation is similar in the field of biowaste management.

Biowaste management in the European Union is regulated by a number of legal acts. Directive 1999/31/EC on the landfill of waste (Dz.U. L 182 from 16.07.1999, p. 1) is such a document that requires Member States to limit the landfilling of biodegradable municipal waste. The purpose of these measures is to reduce the production and emission of greenhouse gases from landfills. The next document on biowaste is the framework Directive 2008/98/EC on waste (Dz.U. UE L 312, 22.11.2008). Pursuant to its provisions, the Member States were obliged to selectively collect biowaste, process it in a way ensuring a high level of environmental protection, as well as to use environmentally safe materials produced from this waste.

Biowaste is a special group among biodegradable waste, defined in the amended Act on waste (Dz.U. 2012, Nr 0, poz. 21; Dz.U. 2007, Nr 39, poz. 251). They mean biodegradable waste from green areas, food and kitchen waste from households, catering establishments, mass caterers and retail units, and similar waste from plants producing or marketing food due to their nature or composition.

1.2. POLLUTION OF SEWAGE SLUDGE AND BIOWASTE WITH TRACE ELEMENTS – HEAVY METALS

A group of particularly harmful elements that are characterized by an adverse environmental and health impact are: cadmium, mercury, arsenic and lead. These elements are among the most toxic (Seńczuk, 2019).

Under certain conditions, chromium, nickel, copper and zinc can have harmful effects on plants or living organisms. The increased content of trace elements in sewage sludge mainly comes from the wastewater in the tanning, painting and metallurgy industries. In addition, these metals originate from domestic sewage, surface runoff and occur in sewage as a consequence of the corrosion of pipes (Gawdzik, 2012).

Trace elements present in sewage sludge and waste occur in various forms: dissolved, precipitated, co-precipitated with metal oxides, adsorbed or associated with particles of biological residues. The chemical forms in which they occur are primarily: oxides, hydroxides, sulfides, sulfates, phosphates, silicates or organic connections in the form of humic complexes (Gawdzik et al., 2010; Latosińska et al., 2010). The chemical form of trace elements present in sewage sludge determines their mobility and the extent of penetration from sludge into groundwater or accumulation in plants after using sludge for soil fertilization (Table 1.1).

Table 1.1

Permissible heavy metal content in sewage sludge intended for agricultural use according to Regulation of the Minister of the Environment of 13 July 2010 on municipal sewage sludge (Journal of Laws of 29 July 2010) (Dz.U. from 29 June 2010)

Metal	The content of heavy metals when using municipal sewage sludge in mg/kg dry solids sludge no more than			
	in agriculture and for land reclamation for agricultural purposes	down reclamation sites on non-agricultural purposes	when adapting land to specific needs resulting from waste management plans, spatial development plans or decisions on building and land development conditions, for cultivating plants intended for the production of compost, for cultivating plants not intended for the consumption and production of feed	Average content heavy metals in stable sediments sewage from municipal sewage treatment plant
Cd	20	25	50	3.6 mg/kg
Cu	1000	1200	2000	378 mg/kg
Ni	300	400	500	24 mg/kg
Pb	750	1000	1500	104 mg/kg
Zn	2500	3500	5000	780 mg/kg
Hg	16	20	25	0.78 mg/kg
Cr	500	1000	2500	31 mg/kg

Sources of the origin of heavy metals in sewage sludge have been known and have long been well identified, and the awareness that these metals in excessive amounts can adversely affect the environment and people is sufficient (Kabata--Pendias, 1999).

1.2.1. CADMIUM (Cd)

The increase in cadmium concentration in soils may be caused by the growing consumption of hard coal and lignite in households and higher consumption of coke in industry. Cadmium enters the human body through the respiratory or digestive systems. It belongs to the elements for which the plant root system does not constitute any barrier. The resorption of cadmium and its compounds from the respiratory system is high and reaches up to 40%. Much smaller amounts of cadmium are absorbed from the digestive tract – about 6%. Greater cadmium resorption occurs with a low content of iron and calcium in the diet. Cadmium in the human body is bioaccumulated for up to 20-30 years. The specificity of cadmium toxicity is the delayed symptoms of poisoning, which applies to both acute and chronic intoxication. In the case of chronic environmental exposure of humans to cadmium compounds, mainly renal, respiratory and skeletal disorders are observed.

The ecological disaster in Japan, caused by eating cadmium-contaminated rice, was the cause of the "itai-itai" disease, which in addition to kidney damage is manifested by softening the bones and increasing their fragility (Seńczuk, 2006).

1.2.2. COPPER (Cu)

Copper as a micronutrient is an essential component of many enzymes and proteins. The main form of toxic active copper is copper ion -Cu^{+2}, although the toxic effects of two ionized copper hydroxides -$CuOH^{+}$ have also been demonstrated. The toxicity of copper in the aquatic environment depends mainly on its alkalinity. Copper is less toxic in strongly alkaline and hard waters, which is due to its lower availability – due to the formation of copper carbonate complexes. Therefore, copper toxicity increases with a decrease in water alkalinity and hardness, pH, dissolved oxygen concentration, chelating agents, humic acid content and suspended solids content.

In a wide range of copper concentrations, weaker fertility, a significant reduction in egg hatching, and a reduction in adolescent growth are observed. As with survival, the largest growth disorders occur in the earliest stages of life. At the tissue level, copper causes changes in blood chemistry, including increases in red blood cells, hematocrit, hemoglobin, plasma glucose, and lactic acid dehydrogenase. As with the combination of various toxins, the occurrence of copper contamination along with other impurities is associated with additional effects. The combination of copper and other heavy metals or xenobiotics may produce a synergistic, antagonistic or additive toxic effect (Atkins et al., 2018).

1.2.3. NICKEL (Ni)

Nickel is a ubiquitous trace metal found in soil, water, air and the biosphere. Nickel is a metal that easily bioaccumulates, due to which its concentration in coal seams is generally significant. Crude oil also contains large amounts of this metal. Since nickel is intensively sorbed by hydrated manganese and iron oxides, its accumulation most frequently occurs in places of these metals. This form of nickel is relatively mobile and available for plants. Nickel creates complex compounds with an organic substance that are stable in wide compartments of soil reaction, and thus can be effectively absorbed by plants. This phenomenon increases the toxicity of nickel, especially within organic and wetland soils. The nickel content in plants depends on the soil reaction. The higher the pH, the lower its content.

Excess nickel causes weakening of photosynthesis, transpiration and leads to disorders in the metabolism of some nutrients. The high nickel content is characteristic for sewage and sewage sludge. Nickel is an element essential for animals, but its excess in feed is harmful because it can cause developmental disorders and a number of other diseases.

Nickel poorly absorbed from food does not accumulate in the tissues of animal organisms. However, at increased doses of this metal, nickel is accumulated in the kidneys and serum, while levels in the liver and heart remain unchanged. Nickel and its salts cause irritation of the conjunctiva, upper respiratory tract mucosa and nasal septum ulceration in humans. In addition, nickel has been proven to be carcinogenic, and its toxicity is mainly associated with occupational poisoning and smoking. Additional sources of nickel in the human diet are fats hardened with the use of nickel compounds and cocoa beans, which are characterized by a high content of this metal (Jakubus and Czekała, 2001).

1.2.4. LEAD (Pb)

Lead is a metal widely used in various industries. Lead belongs to elements very common in nature. It is found in soil, water and in the atmosphere, as well as in living organisms.

The symptoms of lead poisoning are both mental and physiological. There are headaches, a metallic taste in the mouth, nausea, vomiting and stomach aches, as well as anxiety and memory disorders, difficulty concentrating and others. Lead alone accumulates in the liver and bones, damages the kidneys and causes disorders in the synthesis of hemoglobin.

Lead is very widespread in the aquatic environment and soil and has a negative effect on organisms. Soft waters, due to the low alkalinity and buffering capacity of the system, are more dangerous because lead is present in the form of soluble salts. However, hard and highly alkaline waters contain sparingly soluble or practically insoluble lead salts, such as phosphate, sulfate, hydroxide, carbonate and basic carbonate (lead white). Lead in the form of various compounds is present in some industrial wastewater. Compounds of lead in very soft waters are highly toxic to fish. The low concentration of lead salts in the soil promotes nitrification processes, while inhibiting ammonification.

Lead delays growth and clearly limits breathing. Harmful effects depend on the resistance of the plant species, soil characteristics and properties, and the form of the lead compound. Greater damage occurs in acidic soils, and significantly less in alkaline soils.

Microorganisms participating in biological wastewater treatment processes show signs of poisoning already at concentrations exceeding 0.1 mg Pb/dm^3 of wastewater (Atkins et al., 2018).

1.2.5. ZINC (Zn)

Zinc is a common component of the earth's crust. It creates silicon minerals, occurs mainly in divalent form, and forms complex ions in hypergenic and soil environments. All zinc compounds are easily soluble, especially in acidic environments, and the released ions form mineral or organic-mineral connections with high mobility. It undergoes rapid precipitation, mainly in the presence of sulfide ions. Zinc adsorption depends on the pH of the environment. At pH 5.8 it is bound by humic acids, and at lower pH values sorption disappears.

Despite the large migration, zinc penetrates into groundwater in relatively small amounts. The content in groundwater for drinking water is 15 μg/L and does not exceed 80 μg/L. The degree of toxicity of zinc in waters depends on the ionic form and changes under the influence of water hardness and its pH. It is assumed that a concentration of zinc above 240 μg/L may be toxic to aquatic organisms, e.g. salmon. This element is 30 times easier to concentrate in both zoo- and phytoplankton compared to the concentration in water. The permissible content of zinc in discharged sewage is 2 mg/L, and the greatest threat is posed by water discharged from zinc and lead ore mines (Kabata-Pendias, Pendias, 1999).

Zinc in plants is an active component of many enzymes such as dehydrogenases, peptidases, and phosphorylases. It plays an important role in the metabolism of carbohydrates, proteins and phosphorus compounds. It affects the synthesis of auxins and the formation of ribosomes. It affects the permeability of cell membranes, which regulates the proportions of cell components, and also increases resistance to drought and disease.

Excess zinc comes mainly from industrial emissions as dust fallout and through soil pollution and is also found in municipal wastes and sewage. Zinc content in plants in contaminated habitats is very high, especially in leaves and roots.

Symptoms of excessive zinc concentration in plants are chlorotic and necrotic changes in leaves, inhibition of seed growth and germination. Excess zinc by some plants such as soybean, tomato, and cabbage can be bound by phytins in low mobility forms.

1.2.5. MERCURY (Hg)

Mercury is a specific metal that is normally a silvery liquid. This metal is one of the strongest environmental poisons, characterized by enormous mobility in the environment. With the development of industry, mercury began to be used in many technological processes and entered into to the water with wastewater. Mercury is very toxic, both in metallic and inorganic form. Mercury enters the human body through the skin, respiratory system and digestive system. Absorption at low doses contributes to systematic cumulative effects and migration in the food chain. Metallic mercury is biotransformed to persistent and harmful methylmercury, the toxicity of which is much higher than that of other mercury compounds. In terrestrial ecosystems, the bioconcentration of methylmercury is greatly dependent on soil conditions, because soil is the place of its accumulation. Humic substances have a great ability to reversibly bind methylmercury.

Mercury compounds have a negative impact on the work of biological sewage treatment plants due to the inhibition of biochemical processes. Very sensitive methods are available to detect mercury at the microgram level per kilogram of material. However, there are analytical difficulties due to the high volatility of many mercury connections. Therefore, the sample should be stored and then transformed and analyzed in an appropriate form. Mercury can escape from the sample as a result of gas diffusion into the air or penetrate outside the container, and the walls of the container may cause sorption of mercury compounds (Małuszyńska, 2011; Timbrell, 2008).

1.2.6. CHROMIUM (Cr)

Chromium is an essential microelement that is present in active centers of many enzymes. Among the properties of chromium, which determine its impact on the environment, depending on pH, the following should be mentioned: the ability to occur in various degrees of oxidation and the stability of the form of occurrence. Chromium changes in soil are more complex because of the variety of forms in which they occur and the factors that shape them. There are two most common forms of chromium(III) in soil: Cr^{3+} and CrO_2^- Cation Cr^{3+} occurs in an acidic environment, its precipitation begins at pH = 5.5.

Also, chromium(VI) comes in two forms: $Cr_2O_7^{2-}$ and $HCrO_4^-$. Cr(VI) compounds dissolve well in both acid and alkaline soil. Almost all hexavalent chromium is of anthropogenic origin.

Only chromium compounds with zero, second, third and sixth degree of oxidation have biological significance. However, the effects of chromium connections on the third and sixth degree of oxidation are so different that they should be considered separately.

Chromium(III) in nature is in the group of trace elements and belongs to microelements that are necessary for the proper functioning of the body. In the case of trivalent chromium deficiency, symptoms of reduced glucose tolerance, weakness, reduced growth and changes in the circulatory system may appear. Chromium(VI) compounds are more toxic than chromium(III) compounds, as they are closely related to their oxidizing properties, which are the cause of, among others, chronic poisoning.

The formation of stable complexes with proteins, as well as the ability to precipitate proteins, was considered the mechanism of the local and harmful effects of chromium on the skin, nasal mucosa and mouth. Hexavalent chromium compounds, especially those with low solubility, have carcinogenic and mutagenic properties.

Deficiency of this element causes chlorosis and inhibits the development of plants. The demand for zinc to cover the physiological needs of plants is 15-30 ppm (Atkins et al., 2018).

1.3. POLLUTION WITH ORGANIC COMPOUNDS

Organic compounds are a diverse group of pollutants that can accumulate in soil in the event of improper use of sewage sludge. Incomplete degradation of organic pollutants in the natural environment creates the risk of toxic, mutagenic and carcinogenic effects of these compounds. The ability of some compounds to accumulate in animal as well as plant organisms may cause an indirect threat to human health. In addition, the diversity of anthropogenic and natural sources of these compounds means that they can affect living organisms through different routes of exposure.

The most common and monitored pollutants for various elements of the environment are organic in nature and consist of the following:

- pesticides (e.g. aldrin, dieldrin, DDT, HCB, HCH),
- polycyclic aromatic hydrocarbons (PAHs),
- polychlorinated biphenyls (PCBs),
- dioxins (PCDD) (Oleszczuk, 2007).

All listed substances are potentially hazardous to the natural environment and humans and, if they occur in excessive amounts, their presence in sewage sludge and biowaste is therefore one of the potential dangers associated with the use of natural waste, e.g. in agriculture or recultivation.

Some European Union countries have introduced standards for those organic pollutants that have a high frequency of occurrence in sewage sludge. The content of organic pollutants in sewage sludge and biowaste in Poland is less often determined, as this measurement is not required for agricultural or reclamation use (Dz.U. 2015, poz. 257).

1.3.1. CHARACTERISTICS OF ORGANIC COMPOUNDS

The presence of toxic organic substances in sediments and waste was noticed only at the end of the last century (Bernacka, 2000; Hermanowicz, 1999). In most European countries, until recently the content of toxic organic compounds in sludge intended for natural, including agricultural use, has not been limited, assuming that they should not occur in concentrations and biowaste intended for recycling, which may pose a threat to human health and the environment (Bernacka, 2000; Czekała, 2002; Oleszczuk, 2007).

Acceptable contents of such compounds as dioxins, furans, and PCBs have been established in European countries such as Germany, Austria, Switzerland and France. In the Scandinavian countries, the permissible content of such compounds as nonylphenol, toluene, the sum of PAHs, the sum of PCBs (Table 2) applies.

Table 1.2

Permissible organic compounds content in sediments on the example of Sweden (Antonkiewicz et al., 2009)

Type of organic pollution	Maximum content in sediments $mg \cdot kg^{-1}$ s.m.
Nonylphenol	50
Toluene	5
Total PAHs	3
Total PCBs	0.42

Currently, there is no standardization in Poland regarding the abovementioned pollutants in municipal sewage sludge and biowaste intended, e.g. for natural use.

1.3.2. SOURCES OF ORGANIC COMPOUNDS

Organic pollutants due to the persistence and concentration in soil, even in very low concentrations, can have a toxic effect on the environment, as well as human health (Manahan, 2010). These substances, derived from sludge, can enter the food chain in three ways:

- direct uptake from soil or from applied sludge into the soil, e.g. by grazing animals,
- sticking to the surface of plants or dusting of soil fertilized with sediment,
- by the uptake of sediment from plant roots.

1.3.3. POPULAR ORGANIC POLLUTANTS CONTAINED IN SEWAGE SLUDGE, BIOWASTE

1.3.3.1. Pesticides

These substances belong to the group of chemical compounds (natural and synthetic) used to combat organisms harmful to humans, animals and plants. Of the chloroorganic pesticide group, the biggest problems are the residues of DDT (Dichlorodiphenyltrichloroethane) and its metabolites (Oleszczuk, 2007).

Types of pesticides

Pesticides are mainly used as crop protection products against diseases caused by fungi and against pests (e.g. weeds, insects, rodents), whose impact can contribute to a significant reduction in yield or complete destruction of plants. Pesticides are also used as a means to prevent, control or destroy pests of stored food (e.g. in warehouses, granaries), pests of trees and wooden products, as well as substances preventing the invasion of parasites (external and internal) in livestock.

Natural pesticides (biopesticides) contain living organisms or substances of biological origin. They are used to combat diseases, fungi and insects in agricultural, fruit, vegetable and forest crops. These include phytoncides – natural substances secreted by plants that inhibit the growth of microorganisms (bacteria and fungi), e.g. allicin secreted by garlic and onions, and plant substances used against pests of crop plants, e.g. tobacco infusions used to control whiteflies and aphids.

Synthetic pesticides have been produced since the end of the 19th century, most often in the form of powder (for dusting) or in liquid form (for spraying). They are also manufactured in the form of granules and aerosols. There are pesticides that fight: viruses (virocides), bacteria (bacteriocides), fungi (fungicides), algae (algicides), weeds (herbicides), unnecessary shrubs and trees (arboricides, silhouettes), nematodes (nematocides), mites (acaricides), insects (insecticides, including aphids on aphids, larvicides destroying insect larvae and insecticides destroying insect eggs), snails (molluscicides, including limacids acting on nude snails), rodents (rodenticides), moles (talpicides) (Biziuk, 2009).

Pesticides also include:

a. Plant growth regulators (PGRs) – stimulating or inhibiting plant life processes, including agents that remove leaves (defoliants), agents that remove excessive amounts of flowers (deflorants), and agents that dry plants (desiccants);
b. Insect growth regulators (IGRs) – stimulating or inhibiting insect life processes, including antifidants that inhibit feeding and laying eggs;
c. Repellents;
d. Substances attracting pests (attractants), e.g. pheromones;
e. Wood preservatives, incl. against fungi and insects.

Pesticides by chemical composition are divided into:

– **DDT (dichlorodiphenyltrichloroethane, nitrogenox)**; summary formula $C_{14}H_9Cl_5$; weight 354.49 g/mol; very durable, half-life in soil: 2-15 years, in water: 56 days, 28 in the river.

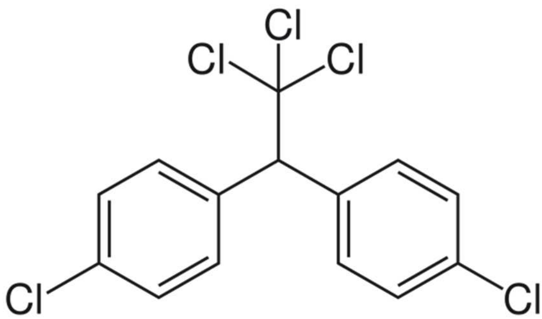


Fig. 1.1. Structural formula of dichlorodiphenyltrichloroethane

An organic chemical compound from the group of chlorinated hydrocarbons. Decomposition products are mainly DDE (dichlorodiphenyldichloroethylene) and

DDD (dichlorodiphenyldichloroethane), which have similar properties and are even more durable. They accumulate in the fatty tissues of animals.

DDT was first synthesized in 1874 by the Austrian chemist Othmar Zeidler. The insecticidal properties of this compound were discovered by the Swiss chemist, Paul Müller, for whom he was awarded the Nobel Prize in 1948. It was widely used at the beginning of the 20th century. It was used on a larger scale during World War II to protect the Allied forces against typhus spread by lice. It seemed to be an ideal plant protection product, used in large quantities around the world in the 1960s.

– **Aldrin (hexachlorohexahydrodimethylnaphthalene)** – an organic chemical compound from the group of halogenated hydrocarbons in the form of a white water-insoluble powder. Summary formula $C_{12}H_8Cl_6$; molar mass: 364.91 g/mol.

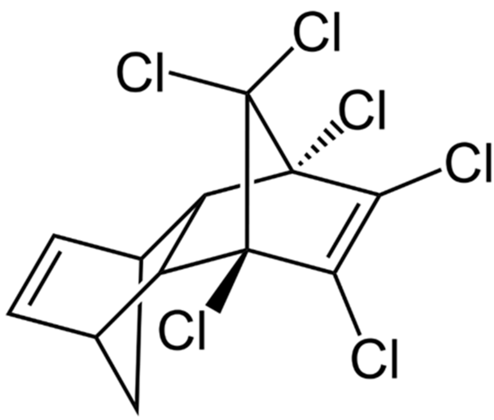


Fig. 1.2. Structural formula hexachlorohexahydrodimethylnaphthalene

Highly poisonous and used as a polychloride insecticide, it is also poisonous for mammals. It is used against grubs and larvae and for soil disinfection. Aldrin is one of the substances considered by the experts to be the "dirty dozen" of the most environmentally harmful pesticides.

– **Dieldrin** – synthetic halogenated insecticide; summary formula $C_{12}H_8Cl_6O$, molar mass 380.91 g/mol.

In the environment or inside the body, dieldrin is formed after the rapid breakdown of aldrin and has a similar chemical structure. It is persistent in the environment and bioaccumulates.

Dieldrin is used in agriculture to protect soil and grain and to control the population of disease-carrying organisms such as tse-tse mosquitoes and flies. It is also effectively used in veterinary medicine as a sheep disinfectant, as a preservative for wood, and it also helps in protecting wool products against moths. Many countries have introduced restrictions or bans on the use of dieldrin. Some countries still allow it to be imported in some instances such as protection against termites.

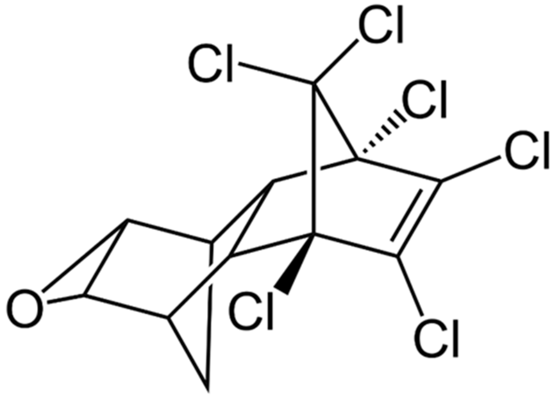


Fig. 1.3. Structural formula dieldrin

Animals and humans may be exposed to dieldrin as a result of ingestion of fish, seafood, dairy products, fatty meats and root crops grown in contaminated soil or water. Dieldrin is highly toxic. Animal studies have demonstrated the destructive effect of this compound by its harmful effect on the liver, central nervous system and immune system. This compound may also interfere with hormonal balance. There is evidence that exposure during pregnancy causes damage to developing fetuses. Dieldrin has very high acute toxicity to aquatic organisms such as fish, crustaceans and amphibians.

– **Lindane (γ-hexachlorocyclohexane)** – colorless crystalline powder, almost odorless (faint musty smell); summary formula $C_6H_6Cl_6$; molar mass 290.83 g/mol.

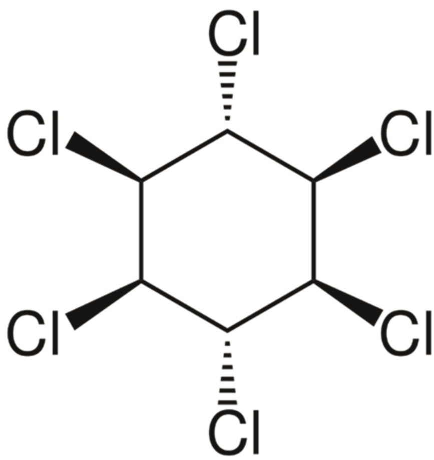


Fig. 1.4. Structural formula γ-hexachlorocyclohexane

As an organic chemical compound, one of the isomers of hexachlorocyclohexane is an active ingredient in pest control preparations, mainly in forestry and industrial crops. It exhibits insecticidal properties (acts as contact and stomach poison), and it was also used to protect herbariums against insects.

In medicine, it is used as a second-line drug for external use against lice and scabies.

These compounds are used to combat insects.

It is characteristic for them that they accumulate in trophic chains, soil and surface waters:

- organic phosphates,
- parathion, malathion; used in pest control in horticulture and agriculture; easily degraded to non-toxic substances;
- synthetic pyrethroids, e.g. allethrin; used to combat insects, less toxic than other types of pesticides.

Biopreparations and living organisms

These are plant protection products based on living organisms (nematodes, bacteria, fungi, viruses). They use natural enemies of pests. An example is the use of a greenhouse phytoseiid mite that feeds on spider mites. Methods based on biopreparations are increasingly used because of the lower risk to human, animal and environmental health.

The importance of pesticides

Benefits of using pesticides:

- increasing the number of crops by reducing the adverse impact of pests;
- limiting the invasion of parasites causing diseases among farm animals; obtaining more animal products (milk, eggs, meat);
- reducing the incidence of infectious diseases among the population (e.g., the eradication of mosquitoes carrying West Nile virus, yellow fever and malaria);
- food protection during storage and transport;
- increasing the durability of industrial products (wood, paper, textiles);
- extending the life of roads by controlling weeds, roadside shrubs and trees;
- preventing excessive algal blooms causing eutrophication of lakes (algicides);
- control of the population of invasive species threatening native species.

Negative effects of pesticide use:

- mutagenic, carcinogenic and neurotoxic effects may contribute to mutations and cancers in organisms directly exposed to them or indirectly as a result of accumulation of pesticides in soil, water, air and food;
- destruction of beneficial organisms in forests, gardens and arable crops (e.g. pollinating insects); 98% of sprayed insecticides and 95% of herbicides affect non-target organisms;
- rapid immunization of pests to the action of a given pesticide, which is associated with the need to use other, stronger measures; groundwater dictionary;
- accumulation in air, soil, surface and underground waters, leading to their pollution;

- accumulation along the food chains of living organisms (the process of biomagnification – an increase in the concentration of a toxic substance in organisms occupying higher trophic levels) (Deshpande, 2002).

1.3.3.2. Polychlorinated biphenyls (PCBs)

Polychlorinated biphenyls are mixtures of congeners (Latin: cum + genus = relative) with different numbers of chlorine atoms and their distribution in the biphenyl molecule. PCBs are products of biphenyl chlorination, which can contain from 1 to 10 chlorine atoms. 209 different congeners with a different degree of substitution are possible (Morrison and Boyd, 2008).

PCBs belong to compounds that do not undergo or slightly undergo biochemical degradation. The lipophilic nature of PCBs as well as high environmental stability and hydrophobic properties favor the accumulation of these compounds in organisms. They accumulate in the fatty tissue of organisms and through the trophic chain can enter the human body and cause damage to the liver, spleen and kidneys. They can damage the immune system and distort the genetic code (Hermanowicz et al., 1999).

PCBs properties such as high temperature stability and resistance to acids and bases have resulted in these compounds finding wide applications in:

- plasticizers and impregnants,
- hydraulic fluids,
- high temperature grease,
- for the production of packaging,
- ink components,
- additives in insecticide preparations (pesticides),
- additives for glues and plastics,
- insulation materials for electric wires, in motors, transformers.

In sewage sludge, the content of polychlorinated biphenyls comes primarily from industrial wastewater. However, PCBs can get into the environment as a result of pouring used oils directly into the soil and water. PCBs also enter the environment as a result of incineration and removal of industrial wastewater and the storage of biowaste. PCBs can also form spontaneously during combustion and chlorination processes of water (Gajkowska-Stefańska, 2001). The presence of PCBs in agricultural sludge will affect the spread of these compounds in the soil environment.

1.3.3.3. Polycyclic aromatic hydrocarbons (PAHs)

Polycyclic aromatic hydrocarbons are compounds containing from two to thirteen benzene rings with any number of alkyl substituents. Generally, PAHs are more lipophilic, less water soluble and less volatile as the molecular weight increases (Morrison and Boyd, 2008). PAHs are products of incomplete combustion of organic substances. The sources of PAH can be natural processes (geochemical processes, combustion processes – burning meadows, forests, biosynthesis and decomposition processes) as well as anthropogenic processes.

Anthropogenic sources of PAHs include:
- production processes (production of plastics, coke, soot, asphalt, aluminum, iron, steel, and catalytic cracking of crude oil),
- carbon coal combustion processes (related to space heating),
- road transport (exhaust gases, abrasion of asphalt surface and tires),
- incineration of organic waste,
- emergency spills of crude oil and liquid fuels,
- cigarette smoke.

PAHs can enter soils from the atmospheric deposit, surface runoff from asphalt roads, organic fertilizers, sewage sludge and composts. The source of PAHs in sewage sludge is industrial sewage. The content of PAHs in sewage sludge can vary. It depends on the type of wastewater, the share of industrial wastewater, the type of sewage system, wastewater treatment processes and sludge treatment (Bernacka, 2000; Hermanowicz et al., 1999).

Mischievousness PAHs

They show strong genotoxic, mutagenic and carcinogenic properties. 16 PAHs are particularly dangerous, with benzo(a)pyrene first. These compounds have relatively low acute toxicity, but very pronounced chronic toxicity. The human body with food takes 3-4 mg of PAH, and the permissible concentration in water is 0.2 mg/dm^3. These compounds are very dangerous because they cause cancerous changes in various tissues.

PAH derivatives with embedded nitrogen atoms (NPAH) are several hundred times more carcinogenic.

Polycyclic aromatic hydrocarbons are classified as persistent organic pollutants that are characterized by a tendency to bioaccumulate and a long half-life in the environment. PAHs consist of two or more aromatic rings. These compounds have different structural forms that are characterized by different mutual positions of benzene rings in the molecule. Some PAH molecules have a so-called “Bay region” – phenanthrene structure. It is an area with increased electron density enabling the formation of, e.g. DNA adducts. The formation of adducts means that these compounds can affect cell replication. Several-ring compounds are classified as acyclic aromatic hydrocarbons. In the environment, 16 PAHs have the greatest toxicity in the environment, such as: acenaphthylene, acenaphthene, anthracene, benzo(a)pyrene, benzo(e)pyrene, benzo(a)anthracene, benzo(b)fluoranthene, benzo (k)fluoranthene, benzo(g,h,i)perylene, benzo(j)fluoranthene, chrysene, dibenzo(a,h) anthracene, fluorene, phenanthrene, fluoranthene, pyrene and indeno(1,2,3-c,d)pyrene (Maliszewska-Kordybach, 2006).

Occurrence in the environment

Polycyclic aromatic hydrocarbons in the environment never occur alone, but always in the form of a mixture. Numerous studies confirm that the presence of one compound from the PAH group in the environmental test is synonymous with the presence of other compounds from this group. These compounds are common, and their main source of emissions are combustion processes in the municipal and housing sector, as well as production processes, especially coke production. In Poland, over 80% of PAHs present in the air are the result of burning fossil fuels. The transport sector is also an important source. Other sources of individual exposure are smoking and some food preparation methods, e.g. smoking, grilling. The best-known aromatic hydrocarbon is benzo(a)pyrene. Due to the strong carcinogenic effect and prevalence, it has been recognized as an indicator of the entire PAH group (Kubiak, 2013).

PAHs are of interest to many scientists due to their genotoxic, mutagenic and carcinogenic properties. Currently, according to the recommendation of the European Commission of February 4, 2005, there is a need for research on levels of compounds classified to 16 PAHs selected by the EU Scientific Committee on Food. The tests should be carried out in particular in selected groups of food products specified in Commission Regulation (EU) No. 835/2011 of 19 August 2011.

The main heat treatment process that contributes to high levels of PAH pollution is smoking. Smoke production is an example of an incomplete combustion process in which many polycyclic aromatic hydrocarbons are produced (Maliszewska--Kordybach, 2006; Kubiak, 2013).

The occurrence of PAHs in all elements of the human environment: in food, air and soil, makes exposure to them common. They get into the human body through various routes: by inhalation, through the skin and while eating food. The dermal route is considered to be the least relevant for environmental exposure. In oral exposure, the highest level of benzo(a)pyrene is determined in foods high in fat (58.2 μg/kg), while the lowest in vegetables (up to 0.48 μg/kg) as well as milk and dairy products (up to 1.6 μg/kg).

Basic sources of PAHs

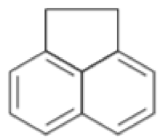
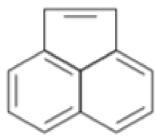
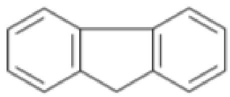
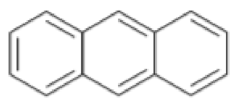
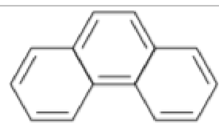
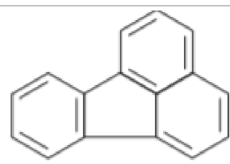
The main and most important source of polycyclic aromatic hydrocarbons are fossil fuels: coal and crude oil as well as asphalt. In addition, they also arise in power plants and combined heat and power plants during energy production. Attention should also be paid to sources of emissions to the atmosphere, i.e. exhaust gases from road transport, fumes from boiler rooms, industrial plants and heating devices. The number of PAHs emitted and their mutual relations are affected by the type of carriers used and the efficiency of the environmental protection devices and filters used. Volcanic activity and forest fires are another source of PAHs. What's more, the cultivation and breeding of animals in biosphere contaminated with polycyclic aromatic hydrocarbons can promote their accumulation in agricultural raw materials and products. PAHs can be taken from soil and water through roots and tubers and as a result of sorption from leaf surfaces (Maliszewska-Kordybach, 2006).

Animals collect PAHs together with plant food and soil during grazing. Polycyclic aromatic hydrocarbons accumulate in adipose tissue. As mentioned earlier, volcanic eruptions, forest fires or hydrothermal processes are natural sources of these compounds. Human activities create additional sources of aromatic hydrocarbons. Food can be contaminated for many reasons, including air, soil and water. This is especially true for vegetables and fruits. Other studies show that food pollution causes traffic. An example is livestock farming near busy roads. As mentioned earlier, the presence of PAHs in food is closely related to the thermal treatment of raw materials, i.e. heating (frying, baking, grilling, smoking), direct drying, oil extraction, or coffee roasting. These are the most important sources of hydrocarbons for edible oils, seeds, coffee, tea, meat and dairy products. The research results of scientists have shown that the higher the temperature and longer process time, the greater the PAH content in the finished product. In addition, smoking additives used to improve the organoleptic quality of products have become a significant source of PAHs.

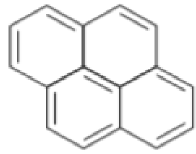
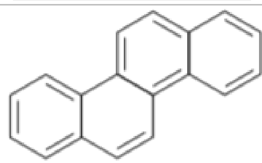
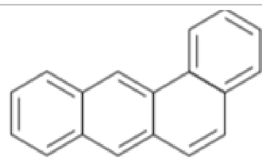
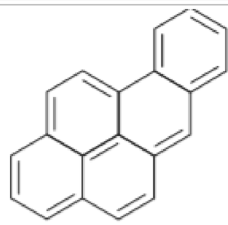
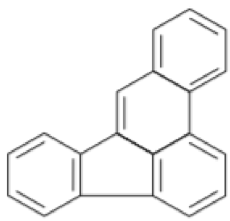
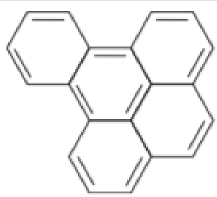
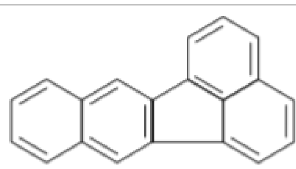
In Table 1.3 are summaried the most common PAHs in the environment.

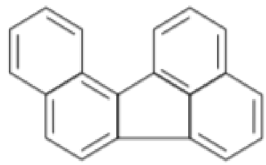
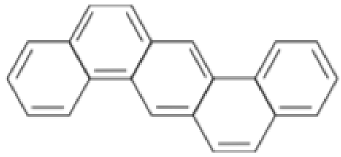
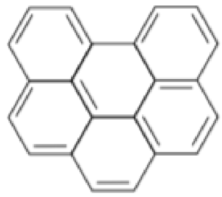
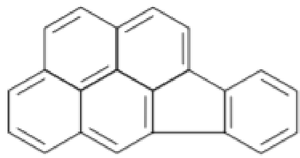
Table 1.3

Polycyclic aromatic hydrocarbons most often determined in the environment and in food (Kubiak, 2013)

Compound	Summary formula	Structural formula	Molar mass
acenaphthene	$C_{12}H_{10}$		154.2
acenaphthylene	$C_{12}H_8$		154.2
fluorene	$C_{13}H_{10}$		166.2
anthracene	$C_{14}H_{10}$		178.2
phenantrene	$C_{14}H_{10}$		178.2
fluoranthene	$C_{16}H_{10}$		278.2

Cont. Table 1.3

Compound	Summary formula	Structural formula	Molar mass
pyrene	$C_{16}H_{10}$		202.3
chrysene	$C_{18}H_{12}$		202.3
benzo(a)anthracene	$C_{18}H_{12}$		228.3
benzo(a)pyrene	$C_{20}H_{12}$		252.3
benzo(b)fluoranthene	$C_{20}H_{12}$		252.3
benzo(e)pyrene	$C_{20}H_{12}$		252.3
benzo(k)fluoranthene	$C_{20}H_{12}$		252.3

benzo(j)fluoranthene	$C_{20}H_{12}$		252.3
dibenzo(a,h)antracene	$C_{22}H_{14}$		278.4
benzo(g,h,i)perylene	$C_{22}H_{12}$		276.3
indeno(1,2,3-c,d)pyrene	$C_{22}H_{12}$		276.3

Effects on the human body

Individual compounds belonging to PAHs have different lithophilicity. This affects the amount of absorption of these compounds in the human body. Polycyclic aromatic hydrocarbons have a negative effect on endocrine, development and reproductive processes as well as are mutagenic compounds (Manahan, 2010).

The most important health effect on the human body is the initiation of cancer by nine compounds from the PAH group. The strongest carcinogens are benzo(a)pyrene and dibenzo(a,h)anthracene. Benzo(a)pyrene has been classified as a proven carcinogenic compound. It is a compound that works without thresholds, i.e. exposure to any concentration of a substance can cause cancer. Exposure to this compound by inhalation creates the likelihood of developing lung cancer, especially for coking plant employees (Bień and Neczaj, 2011; Bernacka, 2000).

Polycyclic aromatic hydrocarbons also pose a risk of premature delivery and fetal growth disorders. These compounds, by binding to the DNA structure of the placenta, show mutagenic effects, which causes the risk of spontaneous miscarriages in early pregnancy. In addition, it has been shown that transplacental transport initiates the formation of oxidative stress negatively affecting the fetal nervous and endocrine systems (Bień and Neczaj, 2011).

Due to the proven negative impact of polycyclic aromatic hydrocarbons on human health, it is necessary to take measures to reduce the exposure of the population to these compounds. Elimination of emission sources would be the most effective preventive measure. The best solution is to introduce changes to the method of heating apartments, because the municipal sector is still the most serious source of PAHs emissions. The fact that benzo(a)pyrene is at a level that greatly exceeds the average normative values in all urban agglomerations and in fourteen voivodships in Poland is currently a cause for great concern (Dz.U. Nr 137, poz. 984; Bień and Neczaj, 2011).

1.3.3.4. Polychlorinated dibenzodioxins (dioxins) and polychlorinated dibenzofurans (furans)

Fires and volcanic eruptions are only responsible for the presence of a small percentage of dioxins in the environment. Although dioxins appear in the environment for natural reasons as well, the largest source of these harmful substances is human activity. Their main source are uncontrolled and unsecured waste incineration processes, such as death utilization in obsolete or technically damaged incinerators and boiler rooms, as well as burning garbage in the open air. The metallurgical and metallurgical industries also have an impact on the concentration of dioxins in the environment.

These compounds known in the environment as secondary products are by--products of combustion with the participation of chlorinated aromatic precursors, especially in conditions of oxygen deficiency. Dioxins contain two benzene rings that are connected by two oxygen atoms, while furans also contain two benzene rings but are connected by one oxygen atom. Dioxins and furans are also produced in the production of organic compounds such as: chlorophenol, PCB, and naphthalenes. The source of dioxins and furans is coal combustion and municipal waste incineration (Laskowski and Migula, 2004; Namieśnik and Jaśkowski, 1995).

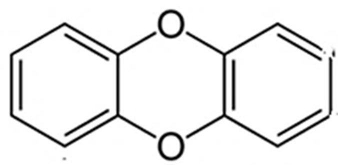


Fig. 1.5. Structural formula of dioxins

Dioxins, together with polychlorinated dibenzofurans and polychlorinated biphenyls form a group of dioxin-like compounds. Due to their chemical structure, all DLCs are hydrophobic. Their solubility in water is almost directly proportional to the number of chlorine atoms in the molecule.

Studies have shown that dioxins and furans introduced into the environment and entering sewage sludge mainly come from the atmosphere as a wet and dry deposit, from transport, the pulp and paper industry, textile leather, oils from metalworking, dry cleaning, households, and wastewater treatment processes (Bernacka, 2000; Oleszczuk, 2007).

The above-mentioned groups of impurities belong to the “most popular” organic xenobiotics, which should be noted due to their high toxicity and persistence in the environment. In Poland, the most common way to utilize sewage sludge is to store it compared to other European Union countries, where over 40% are used for plant fertilization and remediation. As sewage sludge will be increasingly used for fertilizing purposes, there is great danger associated with introducing the above-mentioned groups of organic compounds into the environment, including soil.

1.4. SUMMARY

Neutralization of trace elements and detoxification of organic pollutants in sediments. Increasing the amount of municipal sewage sludge used in agriculture may be associated with the need for sludge treatment. Previous knowledge acquired as part of projects on remediation of soils contaminated with trace elements allows for quite effective selection of natural methods to reduce the negative effects of soil pollution. These methods are relatively cheap and essentially consist in reducing the bioavailability of trace elements.

The widespread occurrence of PAHs and their impact on the human body mean that there is a tendency to develop new methods to reduce their impact on the environment and to conduct thermal processes (in food processing) limiting their formation.

In modern society, there is a growing awareness of environmental protection of interest in food produced without the risk of introducing pollution from both natural sources and those resulting from the specifics of processing in technological processes. Achieving this goal is accomplished through research on the occurrence, formation and impact of undesirable substances in both the human environment and in food, and most importantly, reducing them to a minimum.

The development of new methods of determination and methods of reducing PAHs in food and the environment, and the implementation and compliance with regulations regarding their maximum levels allows the creation of conditions to protect consumer health and reduce the impact of contaminants on human health.

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Chapter 2

Analysis of the suitability of currently used methods for assessing the toxicity of contaminants found in sewage sludge and biowaste

Katarzyna CYGNAROWSKA

2.1. INTRODUCTION

As a result of constant growth of the amount of sewage sludge and biowaste, managing these substances has become a significant ecological problem. The annual production of dry matter of sewage sludge in the European Union is more than 10.96 million tons (He et al., 2014). For many years, sewage sludge and biowaste were mostly stored, burned or composted. Another way of neutralizing sewage sludge and biowaste, which is more economical and often more environment-friendly, is using them as a fertilizer on agricultural soils (Latare et al., 2014). Sewage sludge and biowaste may be rich in organic matter as well as macro and micro elements, so they can serve as an alternative for fertilizers and successfully increase the dry matter yield of various crops (Singh and Agrawal, 2008).

Decisions on how to manage the sludge largely depend on the knowledge of chemical and biological hazards identified in the sludge. Both municipal and industrial sewage can be contaminated with chemicals dangerous for human life and health, which cannot be neutralized as part of processes used in sewage treatment plants and thus concentrate in the produced sewage sludge. One group of such contaminants is refraction compounds (which are hard to decompose or cannot be decomposed biochemically). Among them, especially dangerous are toxic contaminants, causing physiological disorders in plant and animal organisms, and in higher doses, even death. Toxic substances are i.a., heavy metals (e.g., arsenic, copper, lead, cadmium, mercury, zinc, chromium and nickel), polycyclic aromatic hydrocarbons (PAHs), aromatic amines, polychlorinated biphenyls (PCBs), polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDDs/Fs), pesticides, pharmaceuticals and many others (Kapanen et al., 2013). These contaminations have at least one of the following characteristics: carcinogenic, mutagenic or

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teratogenic (causing defects in the development of an embrio or a fetus) activity and the ability to bioaccumulate in the food chain of humans and animals.

The need to manage sewage sludge and reintroduce all the substances occurring in it to the ecosystem generates the need to carry out quantitative and qualitative tests of contamination in sewage and sewage sludge. The development of chemical analysis methods allows us to identify most chemical compounds present in the sludge. However, identification is not enough to answer the following questions:

- What influence can the specific substance (with the specific concentration) have on plant and animal organisms living in the ecosystem?
- How can a substance with a specific concentration affect the human organism directly and indirectly?

Bioindication is the method of assessment of the environmental status that allows us to learn the total toxicity caused to the ecosystem by all the harmful substances taken together. This method involves the use of living organisms present in the environment, called bioindicators, indicator organisms or stenobionts, which have a very narrow range of ecological tolerance to a certain agent (or agents). For example, the presence of crayfish in a lake proves the lack or low concentrations of toxic substances, and the presence of bees in an ecosystem proves the lack or low concentrations of pesticides in the vegetation. When analyzing the total toxicity, we should remember mutual influences of the identified compounds. Different groups of substances present in the environment may interact and thus affect the results of toxicity tests. The interaction may be positive (synergy effect, enhancing the toxicity) or negative (antagonistic activity, reducing the toxicity). The toxicity of the ecosystem as a whole is not equal to the sum of individual toxicities of each substance present in it.

The composition of sewage sludge or biowaste does not provide all the information about its hazardous potential. It is not enough to precisely assess the potential hazard caused by its use for natural or industrial purposes. Objective assessment is possible thanks to toxicological tests, including bioindication assays. Toxicity is a property of a substance causing disorders in biological functions or the death of cells, organs or entire organisms. The effects of toxic activity result from chemical or physicochemical reactions between the toxic substance and the biological system of the organism. Toxicity may be acute or chronic. Acute toxicity tests show the effects substances occurring i.a., in surface waters, bottom sediments, soils, sewage, sewage sludge or biowaste have on organisms. Chronic (long-term) toxicity tests provide information concerning the negative effects of substances on individuals and populations in the conditions of prolonged activity.

The results of toxicity tests obtained for a specific substance using the same species and in the same conditions are not always repeatable. This may be caused by various adaptation factors of organisms, leading to differences in the effects and toxicity levels of particular substances. Therefore, different concepts and measures are used when assessing toxicity. The measure of toxic influence on the organism is the amount of chemical substance causing (or not causing) a biological effect expressed as the proportion of organisms responding to that amount. This value is provided in weight units with reference to the body mass or surface area, and less frequently, to the time of exposure to the toxic substance.

The end point which is the easiest to observe in toxicity tests is the death of the organism as a result of exposure to a lethal dose. In order to be able to compare and evaluate the effects of the tested substances, the concept of lethal dose (LD) has been introduced. LD is the amount of a substance that causes death after one-time administration. Laboratory tests usually determine LD_{50}, i.e., the dose of the toxic substance causing the death of 50 out of 100 tested organisms. For gaseous substances, lethal concentration (LC) is determined. The minimum dose (dosis minima, DM) is also used to refer to the lowest amount of the substance causing the first observable effects of toxic substance activity. Another measure is the effect concentration (EC), causing any changes in tested organisms, e.g., inhibition of growth or biochemical processes. The most frequently determined value is EC_{50}, referring to the concentration of the toxic substance that inhibits the physiological process by 50%. The result is provided with reference to the experiment duration (Walker et al., 2005).

2.2. TOXICITY TESTS

Many sewage treatment plants in Europe are not able to produce environment-friendly sewage sludge that could be used i.a., to fertilize soils (Mininni et al., 2015). The mobility of chemical compounds present in sludge and biowaste and the degree of releasing them to soils depend on a number of factors, such as the pH and chemical composition of the soil, organic matter present in it, redox potential and metal speciation (Malara and Oleszczuk, 2013). The contact of hazardous sewage sludge and biowaste with soils may cause the accumulation of dangerous substances in soil and their transfer to the food chain at different trophy levels (Pathak et al., 2009). Hence, it is necessary to use simple, quick and cheap but also accurate and sensitive analytical strategies of assessing the toxic environmental impact of chemical compounds present in sludge and biowaste. Therefore, a number of biological tests of acute toxicity have been developed to determine the levels of toxicity of those compounds for water and land organisms. The tests involve the use of microorganisms, plants, invertebrates or fish. Currently, several dozen bioindication methods are used to evaluate the impact of potentially hazardous substances on various parts of the ecosystem. These include a number of methods to assess the effects of toxic influences of substances present in sewage sludge and biowaste.

Methods combining chemical analysis of hazardous compounds and biological toxicity tests have proved to be the most effective in identifying the main toxic substances in sludge and biowaste. The use of toxicity identification evaluation (TIE) procedures allows us to obtain information of the most toxic contaminants present in the studied medium and help determine their total share in the general toxicity of the tested sample (Ferraz et al., 2017).

2.2.1. BACTERIAL ASSAYS

Assays based on testing the effects of metabolic activity of microorganisms exposed to substances present in sewage sludge or biowaste are very popular. A number of methods make use of bacteria to test toxicity. The assessment of chemical stress is usually quick, cheap and accurate, which makes it possible to apply that procedure to many samples at a time. There are some standard, widely used tests based on growth inhibition, e.g., the *Pseudomonas* growth inhibition test (Flockton et al., 2019) or the activated sludge test (Friedrichs et al., 2017). Among the most common are *Aliivibrio fischeri* or *Photobacterium phosphoreum* bioluminescence inhibition assays. *Aliivibrio fischeri* is a Gram-negative, rod-shaped bacterium found globally in marine environments. It is bioluminescent and is mostly found in symbiosis with various marine animals. The bacterial enzyme luciferase allows *Aliivibrio fischeri* bacteria to naturally emit light as a result of the following reaction:

$$FMNH_2 + O_2 + R - CO - H \rightarrow FMN + R - COOH + H_2O + LIGHT$$

The intensity of the generated light is proportional to the metabolic state of the cell. Stress conditions caused by the presence of toxic substances have a negative impact on metabolism, thus slowing down the cell activity and reducing the intensity of the generated light.

Since those bacteria live in a water environment, an aqueous solution containing the substances accumulated in sludge or biowaste must be prepared before the experiment. Through the comparison of intensity of light emitted by the bacteria in the tested sample and in the control sample (containing physiological saline solution) we can calculate the percentage of inhibition (I%) using the formula:

$$I\% = [1 - (\text{sample light}/\text{control light})] \times 100$$

The result of the assay is the EC_{50} value, i.e., effective concentration of the toxic substance causing 50% reduction of luminescence (Table 2.1). The EC_{50} values obtained after 15 minutes are converted into toxicity units (TU [g/L]) using the formula:

$$TU = [1 / EC_{50}] \times 100$$

A number of classification scales (e.g. by Persoone, Liebmann) have been developed to determine the class of toxicity. One of them is the system proposed by Persoone, with the following classes depending on the obtained TU value:

- class 0 – $TU = 0$ – no toxicity
- class 1 – $0 < TU < 1$ – no significant toxicity
- class 2 – $1 < TU < 10$ – significant toxicity
- class 3 – $10 < TU < 100$ – high acute toxicity
- class 4 – $TU > 100$ – very high toxicity

Table 2.1

Toxicity values obtained by *Vibrio fischeri* expressed as 50% bioluminescence inhibition (EC_{50}) and toxicity units (TU) (Farré and Barcelo, 2003)

Substance	EC_{50}, µg/mL	TU
Acetaminophenol	173	0.58
Alcohol ethoxylate: $C_{10}EO_x$	3.25	30.76
Alcohol ethoxylate: $C_{12}EO_x$	0.55	182
Benzene sulphonate	1223	0.082
2-Chlorophenol	34.82	2.87
4-Chlorophenol	21.21	4.71
Dichlofluamid	0.136	735.3
2,4-Dichlorophenol	2.85	35.09
Endosulfan	5.63	17.8
Fluorene	4.1	24.2
Ibuprofen	12.1	8.26
Methomyl	1005.89	0.099
Naproxen	21.2	4.76
16 PAHs	0.19	520
Phenantrene	0.13	797
Phenol	7.99	12.5
Polyethylene glycol	127.4	0.79
Sea-nine (antifoulant)	0.0584	1712
2-(thiocyanomethylthio) benzothiazol: TCMTB	0.0268	3731

The luminescence assay has many advantages. It is quick, sensitive, accurate and repeatable. The simplicity of the assay means it can be used to test various environmental samples, including sewage, sewage sludge extracts and biowaste. An ISO standard including high repeatability and simplicity of experiment has been established for the assay based on bioluminescence inhibition (Escher et al., 2017; Di Nica et al., 2017; Rubinos et al., 2014).

2.2.2. BIOSENSORS

Biosensors are analytical devices that can be used to assess the toxicity of the tested sample. They convert biological signals into measurable ones. The output signal is a biological element of the microoorganism, e.g., enzyme (Nguyen et al., 2019) or DNA (Kavita, 2017), which is detected and connected to a converter that converts it to a measurable signal (Table 2.2). Toxicity tests based on bacterial biosensors contain immobilized live bacteria cells. For example, a test based on the inhibition of conductivity of a polymer covered with agarose involves *Saccharomyces cerevisiae* impregnated on an agarose layer. Biosensors can detect various biological variables, e.g., changes in bacterial UV absorption. Other tests are based on changes in cellular respiration, checking the number of electrons produced in this process with the use of a pair of electrodes (a carbon electrode and an Ag/AgCl reference electrode) and an amperometric sensor (Table 2.2).

Table 2.2

Principal transduction systems used in biosensors (Turdean, 2011)

Transduction system	Measurement	Parameters
Electrical	conductometry	conductance
Electrochemical	amperometry potentiometry	current voltage at zero current
Piezoelectric	mass-quartz crystal microbalances mass-surface acoustic waves	mass velocity and so forth
Optical	fotometry fotometry refractometry	luminescence fluorescence refractive index
Thermal	calorimetry	temperature

The CellSense system based on electrolysis can be used to test turbid samples or even suspensions. The measurements are not disturbed by the sample's turbidity, which is definitely an advantage of that technique (Table 2.3).

Table 2.3

Toxicity values obtained using the CellSense biosensor either with *Pseudomonas putida* and *Escherichia coli* expressed in 50% bioluminescence inhibition (EC_{50}) and toxicity units (TUs) (Farré and Barcelo, 2003)

Substance	*Pseudomonas putida*		*Escherichia coli*	
	EC_{50}, μg/mL	TU	EC_{50}, μg/mL	TU
Alcohol ethoxylate: $C_{10}EO_x$ Alcohol ethoxylate: $C_{12}EO_x$	75 69	1.33 1.45	92 596	1.08 0.168
2-Chlorophenol	296	0.34	250	0.4
4-Chlorophenol	239	0.42	201	0.498
2,4-Dichlorophenol	247	0.4	393	0.254
Endosulfan	3.38	29.6	ni	ni
Pentachlorophenol	320	0.31	0.037	2703
Polyethylene glycol 2,4,6-Trichlorophenol	33 256	3.03 0.39	400 0.67	0.25 149.2

ni – not investigated

A disadvantage is the impossibility to test the toxicity of samples including substances that can precipitate on the electrode or remove its bacterial film. This means that many substances, including aggressive solvents, are excluded. Each measurement must be preceded by the test of electrochemical activity of the sample. Sensitivity to some substances and low repeatability of results are the main disadvantages of biosensors based on a system of electrode pairs (Table 2.4) (Farré and Barcelo, 2003; Malhotra and Turner, 2003; Turdean, 2011).

Table 2.4

Advantages and disadvantages of a whole cell-based biosensor (Turdean, 2011)

Advantages	Disadvantages
– more sensitive and detailed than chemical methods, – produces real-time data and can be applied in field work or in situ analysis, – fast, less expensive, and less intensive labour, – cheaper to use because the active biological component does not have be isolated and because microorganisms are living, unlimited quantities can be prepared relatively inexpensive, – react only to the available fraction of metal ions, – does not involve specialized training and the bulky, fragile equipment	– short lifetime, – conditions (reagents, incubation time, pH, temperature) can effects the biosensor performances, – limited of selectivity, – lack of genetic stability

2.2.3. TESTS USING INVERTEBRATES

The presence of earthworms is one of the most important factors contributing to soil loosening and fertilizing. They live in all types of soil. There are approx. 800 worms in various stages of development in 1 m^3 of soil. These organisms fragment and mineralize organic matter, changing the structure and chemical composition of soil. Therefore, they are very susceptible to the activity of all substances present in the soil, especially the toxic ones, which can be introduced to the soil with hazardous sewage sludge or biowaste (Babić et al., 2016; Kinney et al., 2012). *Eisenia fetida* is a species used in biohumus production and in waste disposal processes. It can live up to 15 years long: the longest out of the several hundred described earthworm species, whose life cycle is on average 4 times shorter. This species reproduces quickly and is highly sensitive to many toxic substances occurring in soils. Because of its advantages, it is often used in ecotoxicological tests. *Eisenia fetida* is one of the best organisms for ecotoxicological assays carried out with the use of soils and solid waste for which OECD and ISO toxicity standards have been established (Nahmani et al., 2007). Experiments using earthworms can be used to assess both acute and chronic toxicity. Chronic toxicity caused by the conditions of long-term exposure to the tested substance or group of substances present in soil or sludge is most often tested. The expected measurable and directly observable effect of such experiments is usually the reduction in biomass growth, increased mortality, color changes, mobility problems and problems with reproduction (a lower number of cocoons)

(Babić et al., 2016; Molina et al., 2013; Xing et al., 2014). The toxic effect can be assessed, not only through direct observation but also, i.a., through measurements of morphological changes, multixenobiotic resistance mechanism (MXR) activity or lipid peroxidation levels (the biological process of lipid oxidation leading to the formation of lipid peroxides). The standardized MXR test is based on the measurement of model fluorescent dye concentration in earthworm bodies. Even short-time exposure to a toxic substance results in observable MXR inhibition, whose degree is proportional to the amount of the stressor (Babić et al., 2016).

Apart from direct observation of the influence of hazardous substances on the development of *Eisenia fetida*, histopathological assays (microscopic identification of lesions occurring in organisms' cells and tissues) are also performed. Feeding on organic substances present in soil, earthworms absorb them via their alimentary system, which leads to their exposure to direct contact with toxic substances occurring in it. Organisms exposed to the activity of substances present in soils for a longer period of time also absorb them indirectly, via their thin, semipermeable epidermis. Histopathological assays allow i.a., for the identification of physiological changes in earthworms' organs and tissues and determine the impact of hazardous substances on the thickness of body walls, mid-gut epidermis injury, and the area of gut resorption (the process of substance transpiration through surfaces) (Babić et al., 2016; Christofoletti et al., 2012).

2.3. SUMMARY

The discussed methods of ecotoxicological assessment of contamination in sewage sludge and biowaste are commonly used all over the world. All the assays involve living organisms, but none of them is universal enough to be undeniably better than the others. The main reasons for this are the kind of microorganisms used and the limitations of particular methods. Each microorganism used in the tests has a different specificity. It may be very sensitive to some groups of toxic substances but resistant to others, even if only to a small extent. The compounds may be widely occurring in the environment, very rare, or even completely new, not yet tested for their negative effect on the ecosystem. Therefore, in order to determine the degree of toxicity of a substance and its dangerous dose, we should perform tests involving various organisms living in the given environment. Different species display different levels of resistance to the same substance. Bacteria and invertebrates are organisms that develop most frequently in sewage sludge and biowaste which can enrich soils in valuable organic and mineral matter, so the available assays most often involve some of these.

There are at least several automated devices on the global market whose operation is based on the analysis of intensity of light generated by luminescent bacteria, e.g., Tox-Alert from Merck, Microtox from Azur Environmental or LUMIStox from Beckman Instruments (Farré and Barcelo, 2003). Figure 2.1 shows the Microtox system used for tests at the Czestochowa University of Technology.

Notwithstanding all its advantages, the test using *Vibrio fischeri* has some limitations. The maximum methanol concentration tolerated by luminescent bacteria is 10%. Moreover, these bacteria are marine microorganisms, so in order for the assay to be accurate, filtration in a saline solution is necessary before each test. The salinity of the sample reduces the solubility of some organic substances, which causes the turbidity of the studied solution. Despite these limitations, assays based on the analysis of bacteria bioluminescence have for many years been among the quickest and most effective tests to assess toxic influence on the environment.

The main advantages of biosensors are the possibility of mass production, online readings, quick reaction and simplicity of application. Methods used to create biosensors allow us to select the best test species for the assessment of toxic effect caused by particular substances occurring in the sample. The most important is the proper choice of microorganisms that will be sensitive enough to detect the expected groups of hazardous substances. Another factor that plays a role is whether the microorganisms can be immobilized. Furthermore, the appropriate measurable biological signal should be selected. As already mentioned (Chapter 2.3), due to the processes occurring in the device, not all substances can be tested using this technique (Farré and Barcelo, 2003).

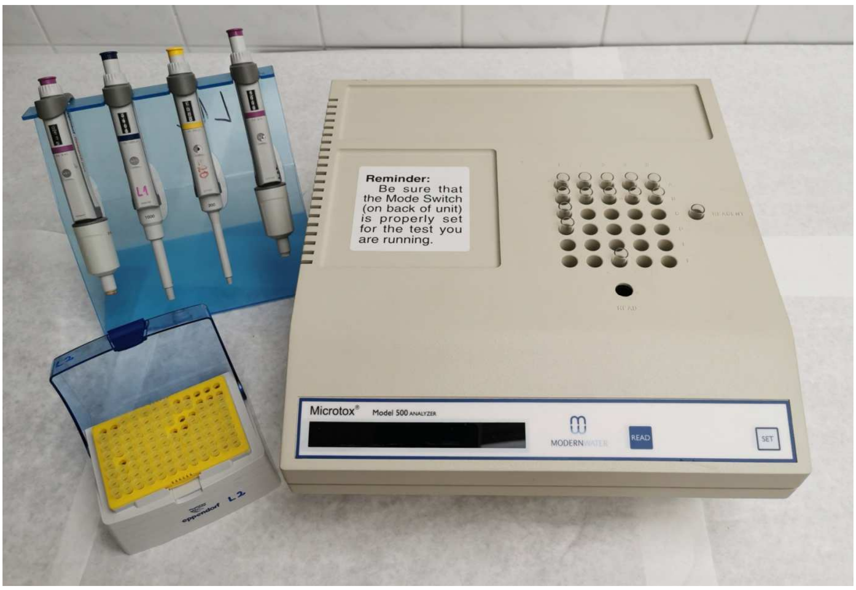

Fig. 2.1. The Microtox M500 system used for tests at the Czestochowa University of Technology

Because they allow for testing various physiological factors and perform morphological, histopathological, and even behavioral analyses, biological assays using earthworms allow to identify precisely (much better than do tests using microorganisms) the mechanisms of activity of hazardous chemical substances present in sludge and biowaste. In addition, the life cycle of *Eisenia fetida* makes it possible to perform short- and long-term tests of acute and chronic toxicity. However, such tests have a lower automation level, are more expensive, more labor-intensive, and take much longer than do experiments with the use of microorganisms (Babić et al., 2016).

Each of the discussed methods of assessing the toxic effect has its limitations, which make it impossible to obtain reliable results for some groups of substances. Still, they are all quick, simple and cheap techniques. The use of various test species provides exhaustive information on environmental hazard, and the combination of these methods with chemical analysis of the tested substances is a good approach to the identification of the most toxic fractions and hazardous compounds present in sludge and biowaste.

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Part II

Methods used for treatment and final disposal of biowaste and sewage sludge

Chapter 3

Targeted recovery of energy and matter: review of technologies for the recovery of matter from biowaste sewage sludge

Ewa SIEDLECKA

3.1. TECHNOLOGIES FOR THE RECOVERY OF MATTER FROM BIOWASTE

Biowaste is defined as biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants. It does not include forestry or agricultural residues, manure, sewage sludge, or other biodegradable waste such as natural textiles, paper or processed wood (Directive 2008/98/EC). Biowaste that is not managed is a serious threat to public and environmental health because it affects olfactory nuisance, attracts insects, rodents, and produces leachate that may contaminate surface and groundwater supplies (Reddy and Nandini, 2011). In addition, uncontrolled disposal of biowaste emits methane, the main greenhouse gas. Biowaste accounts for around 20% of the waste produced in the EU and can serve as a potential resource of valuable chemical compounds, supporting a circular economy. Biowaste treatment in a circular economy solves the problem of resource scarcity, for example, the depleting nutrient stocks such as phosphorus (Zabaleta and Rodic, 2015).

Biowaste treatment technologies include processes that transform discarded biowaste into new products with some value. They can be very simple such as using the stream as an animal feed or very complex, as in the case of extracting chemicals from the feedstock. They can be divided into three broad groups:

- direct use: unaltered or slightly altered, e.g. for land and feed application,
- material recovery: biochemical or chemical extraction and conversion of the biomass into other useful products such as platform chemicals, solvents or fertilizers,
- energy recovery: burning biomass or biogas originating from e.g. anaerobic digestion in order to recover part of its energy contents (Six et al., 2016).

Figure 3.1 presents the most important techniques of matter and energy recovery from biowaste, products and possibilities of their use.

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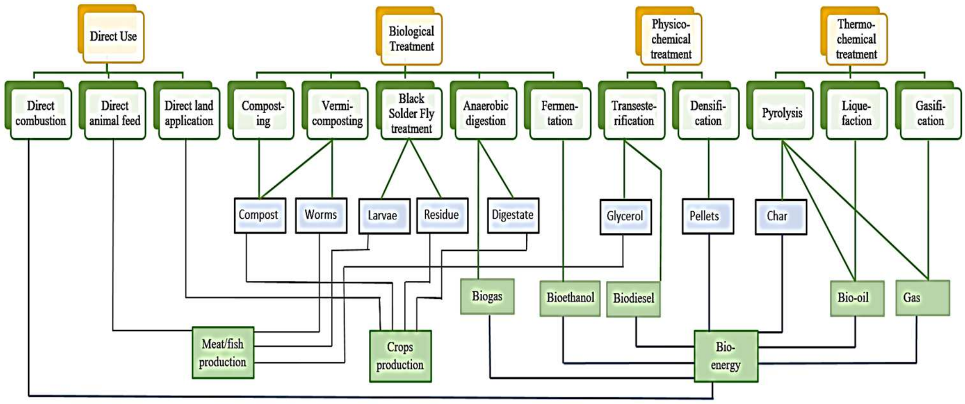

Fig. 3.1. Biowaste matter and energy recovery techniques (based on Lorhi et al., 2017)

3.1.1. DIRECT USE OF BIOWASTE

This method includes direct land use and waste fed directly to animals. The risk of such practices depends on the composition of the biowaste (Lorhi et al., 2017).

- **Direct land application** is associated with the spreading of raw organic waste in the field, which then undergoes natural aerobic biodegradation. Degradation mobilizes nutrients and increases the soil's organic content. This method can have a negative effect on plants and soil. Since the waste probably contains some level of pathogens or trace elements, they can accumulate in plants and soil. This can cause health risks due to food contamination or watercourses as a result of runoff.
- **Direct animal feed** is a simple way to recover value from biowaste. In some countries such as South Korea, Taiwan and Japan, 38.4, 22.1 and 11.5% of biowaste respectively, is processed into swine, poultry and fish feeds to partly substitute the conventional feed ingredients. The biowaste may be treated, such as grinding or drying, and can then be fed in pure form to animals or in mixed form with other feedstuffs. Completely rotten items should not be used for animal feed. When biowaste contains meat or has been in contact with meat, there may be a risk of infection to animals that can then transmit diseases to humans (e.g. Salmonellosis) or other animals (e.g. swine fever or bovine spongiform encephalopathy, BSE) (Lorhi et al., 2017).
- **Direct combustion** (known as open combustion) is a technique where biowaste is burned in the open air or in the presence of excess air. In this process, the chemical energy of the biomass will be converted into gases (Lam et al., 2019). Direct combustion of biomass or biowaste (for example on a grate) is not acceptable due to different pyrolysis and oxidative pyrolysis times for various materials present in the stream waste. Most biowaste can be converted into fuel by gasification because the process is generally more efficient and cleaner than direct combustion (Stąsiek and Szkodo, 2020).

3.1.2. BIOLOGICAL TREATMENT PROCESSES

The process are defined as controlled waste transformation by living organisms. Biological processes – composting and fermentation – have a significant role in closing the waste cycle (Jędrczak, 2018). Important processes for the recovery of matter from biowaste are vermicomposting, Black Solder treatment (BSF), and anaerobic digestion (AD).

- **Composting** refers to controlled decomposition of biodegradable materials under aerobic conditions that results in a relatively stable organic end product called humus (compost) (Fig. 3.2).

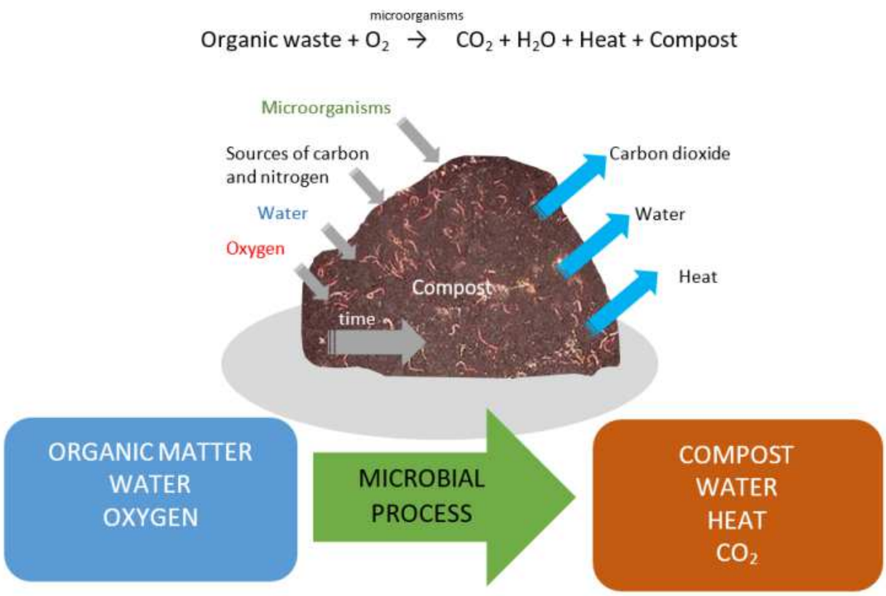

Fig. 3.2. Diagram of composting process

Many different types of biowaste are suitable for composting, that include yard waste (branches, leaves, grass), food waste, agricultural waste and manure. Mixed municipal waste may also be composted, but this is not recommended as the resulting compost quality will be low. The main output product from composting is compost. Besides compost, other output products emitted during the composting process are leachate, water vapour and carbon dioxide (Lorhi et al., 2017). There are many factors that control the composting process, among them the most important are: moisture content, nutrition, temperature and oxygen (El-Haggar, 2007) (Table 3.1). Composting biowaste can be carried out by using different technologies and mechanization and at different scale (Misra et al., 2003) (Fig. 3.3).

Table 3.1

Factors affects the composting process

Factors in composting	Impact on process
Moisture content	• The initial moisture content: 40 to 60% • The ideal percentage of the moisture content is 60%; if the moisture content decreases less than 40%, microbial activity slows down and becomes dormant; if the moisture content increases above 60%, decomposition slows down, and odour is emitted
Nutrition (C:N ratio)	• Microbes work actively if the carbon: nitrogen ratio is 30:1 • If the carbon ratio exceeded 30, the rate of composting decreases • Decomposition of the organic waste material will slow down if C:N = 10:1 or 50:1
Temperature	• The degradation by composting proceeds through three phases: mesophilic stage which lasts for a couple of days, thermophilic phase (up to 70°C), cooling and maturation phase (>15°C) • The ideal temperature range within the compost for it to be efficient varies from 32 to 60°C • The increase of temperature while composting above 55°C, kills weeds, ailing microbes, and diseases including *Shengella* and *Salmonella* • In winter, the composting process is slower than in spring and summer
Oxygen (aeration)	• Continuous oxygen supply via aeration is a must to guarantee aerobic fermentation • Proper aeration is needed to control the environment required for biological reactions • Techniques used to perform the required aeration in accordance with composting techniques: natural composting, forced composting, passive composting, and vermicomposting
pH of material	• Optimum pH ranges between 6.5 to 7.5
Porosity	• Spaces between particle enable supply of oxygen
Toxic substances	• Heavy metals are toxic to thermophilic bacteria

- **Vermicomposting** – the term “vermi” in vermicomposting is derived from the Latin word “vermis” which means a worm. The vermicomposting refers to a composting process that is done by epigeic, anecic or endogeic earthworm species that have a natural ability to colonize and degrade organic wastes. Among the widely used earthworms for vermicomposting are *Eisenia fetida, E. andrei, Eudrilus eugeniae,* and *Perionyx excavates* (Mupambwa and Mkeni, 2018). It is an environmentally acceptable means of converting waste into nutritious compost (known as vermicompost) for crop production (Malińska et al., 2017; Muralikrishna and Manickam, 2017). The texture of vermicompost is fine and heavy metals accumulate in earthworm bodies. Earthworms can process household waste, organic municipal waste, sewage sludge and organic waste from various industries (such as paper, wood and food). Earthworms do not tolerate food waste in the form of dairy products, meat and fish waste, fats and oils, and salty food (Lorhi et al., 2017). Table 3.2 presents the most common systems of vermicomposting and Table 3.3 shows factors with optimal range for vermicomposting.

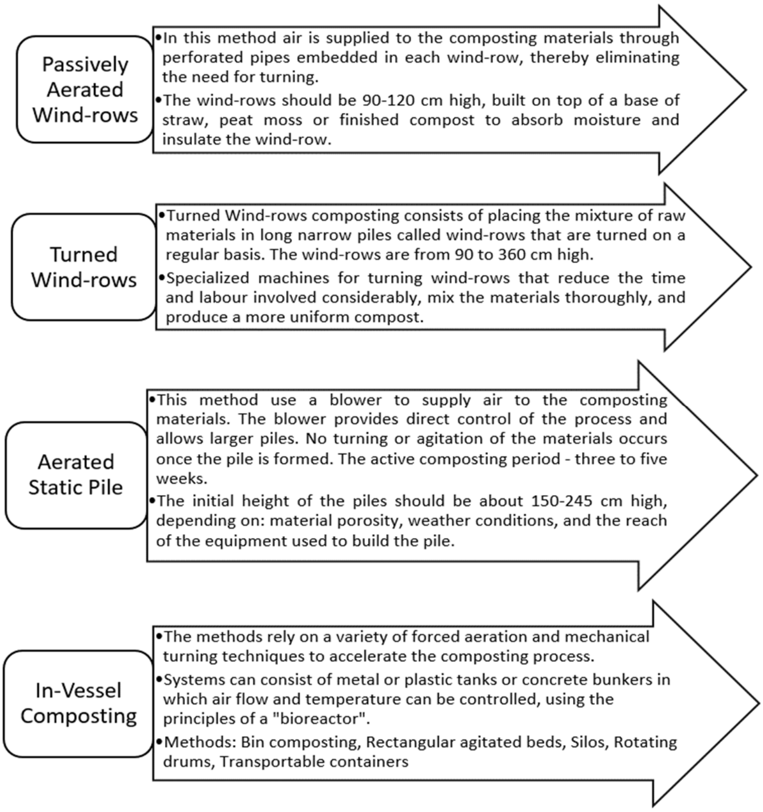

Fig. 3.3. Medium and large-scale composting methods

Table 3.2

Vermicomposting systems (Hanc and Pliva, 2013)

Vermicomposting system	Characterisation
Small-scale domestic systems	Systems consist of a suitable container, bedding, earthworms, and proper environmental conditions
Low technology vermicomposting systems (known as windrow and batch systems)	The system requires a large area of land and is relatively labour-intensive. Since this is usually an outdoor process, there is evidence that a large proportion of the essential plant nutrients are either washed out of the organic matter or can volatilize from it during this long processing period
Medium- and high-technology vermicomposting systems	Systems are represented by manually operated or fully automated continuous-flow vermicomposting reactors. The earthworm populations in reactors reach equilibrium and can usually be run trouble-free without adding or removing earthworms for many years

Table 3.3

Factors with optimal range for vermicomposting (Ali et al., 2015)

Parameters	Optimal ranges for earthworms growth and cocoon production
Stocking density	27-53 worms per kg and 4-8 worms per g/feed
Temperature	25-37°C
Feeding rate	1.25 kg feed/kg worm/day
Moisture	65-70%
C:N ratio	25
pH	4.2-8.0

- **Black Solder Fly treatment (BSF)** is a new technology for processing organic waste. It involves the transformation of biowaste into insect protein and insect oil (Lorhi et al., 2017). The Black Soldier Fly, *Hermetia illucens*, is of the dipteran family *Stratiomyidae*. It can be encountered in nature worldwide in the tropical and sub-tropical areas (Dortmans et al., 2016). Waste biomass is converted into larvae and residue. The larvae consist of ±35% protein and ±30% crude fat. This insect protein is of high quality and is an important feed resource for chicken and fish farmers. Feed trials have confirmed it as being a suitable alternative to fish meal (Rindhe et al., 2019). Other possible products to be explored are the production of biodiesel from larvae or the use of the chitin and the oil. The residue still contains valuable nutrients and might be used as a soil amendment (Lorhi et al., 2017). BSF larvae can compost various types of organic waste or biomass (Fig. 3.4) (Singh and Kumari, 2019).

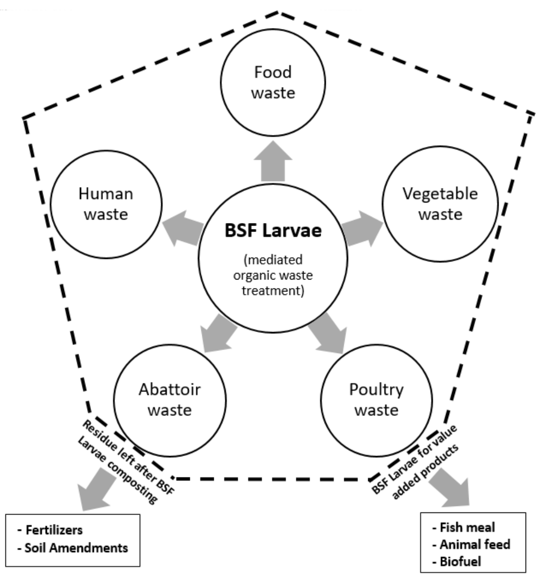

Fig. 3.4. BSF technology (based on Singh and Kumari, 2019)

- **Anaerobic digestion (AD)** is a microbiological process in which organic matter decomposes in the absence of oxygen. The AD process is used to process organic biodegradable matter in airproof reactor tanks called digesters for the production of biogas (Fig. 3.5). The process is generally carried out in four stages: hydrolysis, acitogenesis, acetogenesis, and methanogenesis. To achieve this sequence of four steps, various bacteria (e.g. fermenting, acetogenic and methanogenic bacteria) need to work together. Biomass suitable for digestion is called substrate or feedstock (Table 3.4). Various groups of microorganisms are involved in the anaerobic degradation process which generates two main products: energy-rich biogas and a nutritious digestate (Vogeli et al., 2014).

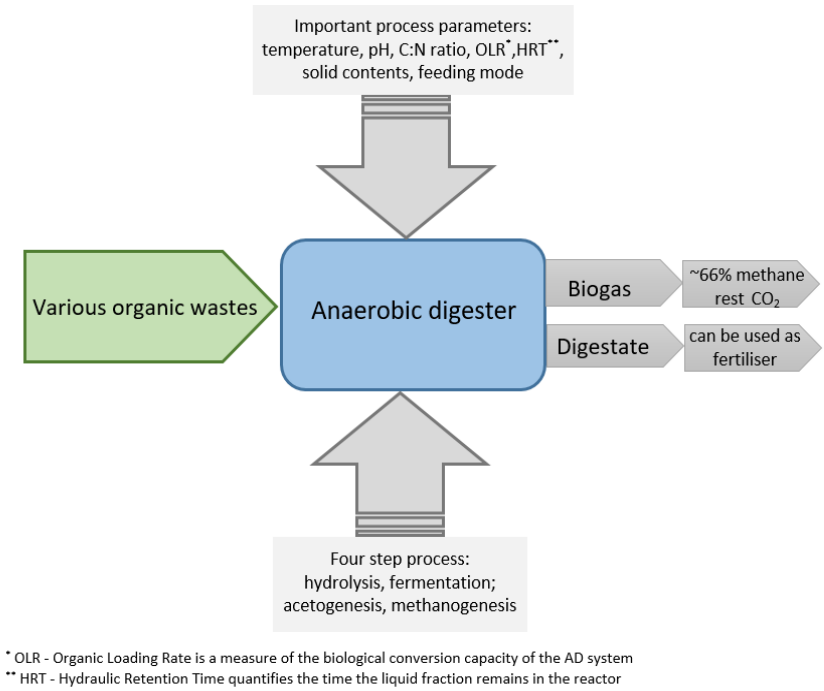

Fig. 3.5. Diagram of anaerobic digestion

Table 3.4

Anaerobic digestion feedstock (Vogeli et al., 2014)

Municipal	**Agriculture**	**Industry**
– Organic fraction of municipal solid waste – Human excreta	– Manure – Energy crops – Algal biomass – Agro-industrial waste	– Slaughterhouse waste – Food processing waste – Biochemical waste – Pulp and paper waste

The AD systems are constituted from reactors to perform a series of bi-metabolism steps. Reactors are the place where growth factors (temperature, pH, nutrients) and operating parameters (retention time, and organic loading rate) are controlled. The digestate is also called effluent in wet systems. Commonly, the effluent from wet-fermentation biogas plants is a very liquid slurry. The effluent from household digesters treating kitchen waste is a good organic fertiliser. However, if the feedstock contains human excreta, the quality of effluent as it leaves the digester is not suitable for direct reuse or discharge. In such situations, a post-treatment step of the effluent is necessary before safe reuse or discharge. The effluent from AD is a good fertiliser in terms of its chemical composition. All plant nutrients (nitrogen, phosphorous, potassium, trace elements) essential to plant growth, are available in the substrate. If the digestate is in solid form (e.g. after dry digestion or after drying of effluent), the suitable post treatment option is composting. High temperatures during the composting process cause one to obtain a hygienised product (Vogeli et al., 2014).

- **Fermentation** is a process of controlled degradation of biodegradable materials under anaerobic conditions (in a closed reactor) at temperatures suitable for mesophilic or thermophilic bacteria. There are two main products of the methane fermentation process: digestate that can be used as a soil conditioner and biogas that can be burned to produce renewable energy or purified and used as fuel for vehicles. The fermentation process is not suitable for the processing of wood-based materials. It is also a more complex technical process and therefore, more expensive to operate than the composting process. Fermentation of biowaste compared to composting seems to be a more favourable solution, both for technical and technological reasons, as well as for economic reasons. In the composting process, the greater part of the energy potential of raw materials is released in the form of waste heat, whereas with anaerobic digestion, over 80% of energy goes into biogas and can be used. Fermented waste, after aerobic stabilization and possible separation of hard parts, is a high-quality product for agricultural management (Jędrczak, 2018).

3.1.3. PHYSICO-CHEMICAL TREATMENT

Methods refers to conversion processes induced by chemical reactions or by applying physical and mechanical force.

- **Transesterification**. In order to obtain the product in the form of biodiesel, vegetable oils or animal fats are subjected to a chemical reaction called transesterification, also called alcoholysis. The process requires a catalysed oil or fat reaction in the presence of alcohol (Fig. 3.6). Fatty acid methyl esters (biodiesel) and waste glycerol are formed (Baskar et al., 2019). Potential feedstocks for biodiesel production from biowaste are waste cooking oil, animal fats from slaughterhouses, and grease from grease traps, typically collected in the septic tanks of restaurants. Glycerol as a by-product has been the subject of many studies. Solutions for the valorisation of glycerol are the following (Lorhi et al., 2017): a) microbially conversion to valuable chemicals using various bacteria, yeast, fungi, microalgae; b) feed ingredient for animal; c) substrate or co-substrate in anaerobic digestion; d) ethanol production; e) microbial fuel cells to generate electricity.

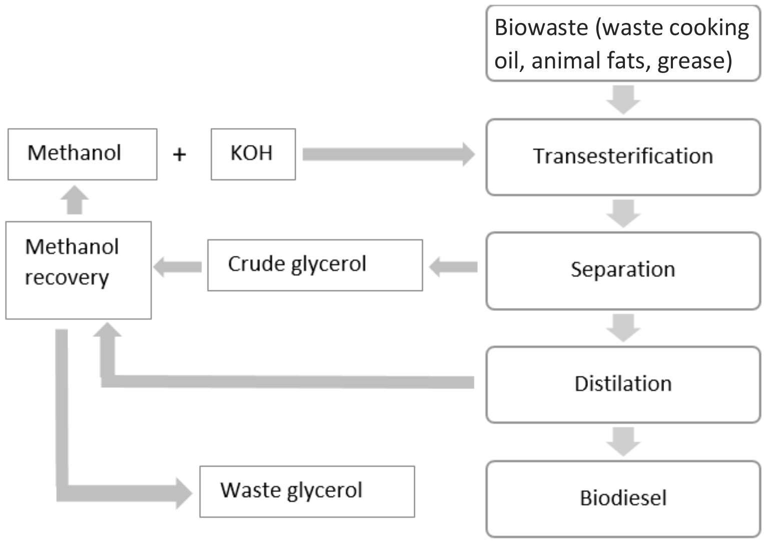

Fig. 3.6. Scheme of biodiesel production (Li et al., 2013)

- **Densification** – process of compaction of the biomass by applying mechanical force or sometimes binding agents to create inter-particle cohesion, resulting in homogenous briquettes or pellets with consistent shapes and sizes, and bulk densities ranging from 450 to 700 kg/m^3. Densification is applied to raw biowaste, as a pre-treatment step for biomass pellet/briquette use in pyrolysis, gasification and combustion systems, and also in the post-processing step for char, that is formed during slow pyrolysis. The resulting char-briquettes are suitable for use as cooking fuel (Kaliyan and Morey, 2010). Biowaste used for densification: crop wastes (paddy straw, bean straw, soya straw, maize straw and wheat straw) and agro-industrial residues (rice husk, coffee husk and soybean husk, bagasse, sawdust) (Lorhi et al., 2017).

3.1.4. THERMOCHEMICAL TREATMENT

Treatment is the decomposition of organic components in the biowaste using heat. Conversion by means of thermochemical technology comprises pyrolysis, gasification and liquefaction (Lee et al., 2019). Figure 3.7 include the most important information about processes.

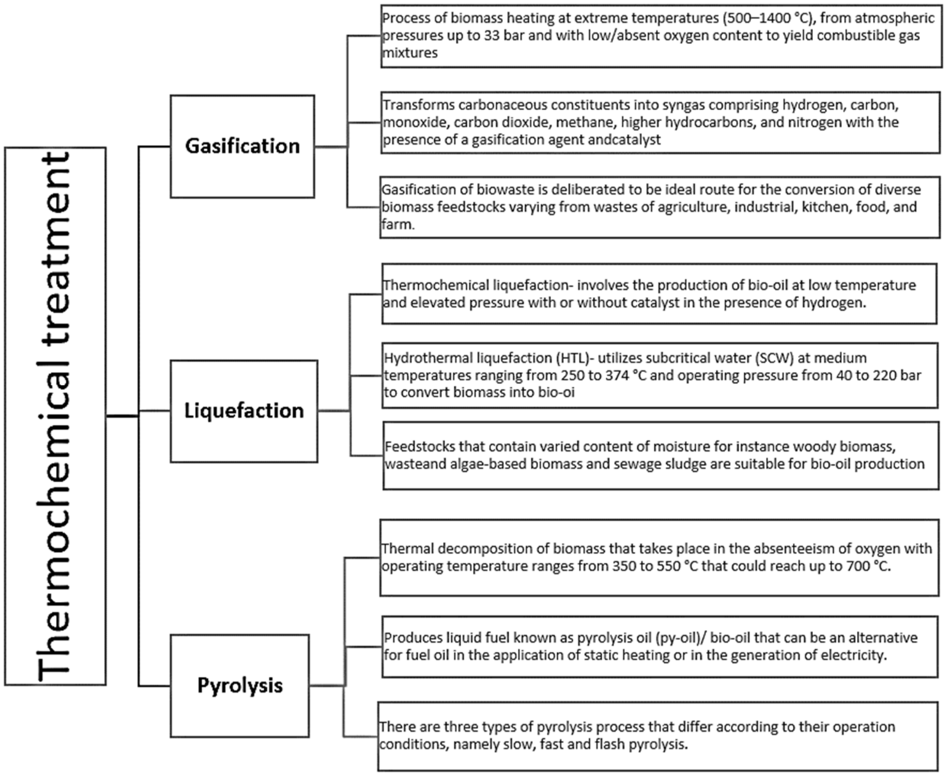

Fig. 3.7. Techniques of thermochemical treatment (based on Lee et al., 2019)

3.2. TECHNOLOGIES FOR THE RECOVERY OF MATTER FROM SEWAGE SLUDGE

Sewage sludge is a kind of waste coming from municipal wastewater treatment plants. Effective treatment of municipal wastewater leads to the formation of large amounts of sewage sludge. Sewage sludge management is an important issue in any modern municipal sewage treatment plant (Grobelak et al., 2019). Sewage sludge management strategy in the EU has two main trends. The first is heat and electricity production (from biogas) as specific products that have renewable potential. The second trend is to treat bio-solids as products in which sewage sludge is the main ingredient (Kacprzak et al., 2017; Przydatek and Wota, 2020).

There are different types of sewage sludge having different physical and biological properties. Typically, sewage sludge consists of primary and secondary sludge.

Primary sewage sludge is taken from the primary settler. It is generated through a mechanical (screening, grit removal, sedimentation) wastewater treatment process. Their amount depends on the retention time and the volume of the settling tank.

Primary sewage sludge usually contains from 93 to 99.5% water, high ration of suspended and dissolved organic matters.

Secondary sludge often called as waste activated sludge is taken from the clarifier (secondary settler). It is generated during biological treatment of the wastewater and contains mainly microbial cells that are complex polymeric organic materials. Waste activated sludge consists of 59-88% (w/v) of organic matter. A small part of the sludge is solid matter in which over 95% is water. The organic matter contains 50-55% carbon, 25-30% oxygen, 10-15% nitrogen, 6-10% hydrogen, 1-3% phosphorus and 0.5-1.5% sulphur. In the composition of the ash from waste sludge, there are mainly minerals such as quartz, calcite or microline. These minerals are formed by elements such as Fe, Ca, K and Mg. Additionally, some heavy metals such as Cr, Ni, Cu, Zn, Pb, Cd and Hg can be found in the sludge (Bień and Wystalska, 2007; Tyagi and Lo, 2013).

3.2.1. SEWAGE SLUDGE AS A RESOURCE

There are two components in sewage sludge that are feasible to recycle: nutrients as nitrogen (N) and phosphorus (P) and energy (carbon). Within sewage sludge there is a considerable amount of nutrients, mainly P and N. Other resources include the reuse of sludge for construction materials, heavy metals, poly-hydroxyalkanoates (PHA), proteins, enzymes and VFA. Table 3.5 presents an overview of resource recovery products from sewage sludge, their typical values and uses. In addition, there are technologies for innovative products from treated sewage sludge: VFA, polymers and proteins in the form of worms, larvae and fungi (Healy et al., 2015).

Table 3.5

Resource recovery products from sewage sludge (Healy et al., 2015)

Products	Typical values and uses
Nitrogen	2.4-5 (% TS)
Phosphorus	0.5-0.7 (% TS)
Heavy metals	Typical recovery values: Ni 98.8%, Zn 100.2%, Cu 93.3%
Construction materials	Dried sludge or incinerator ash. Biosolid ash is used to make bricks
Bioplastic	Microorganisms in activated sludge can accumulate PHA (polyhydroxyalkanoates) ranging from 0.3 to 22.7 mg polymer/g sludge
Hydrolytic enzymes	Protease, dehydrogenase, catalase, peroxidase, α-amylase, α-glucosidase
Biofuel	Syngas, biodiesel, biooil

3.2.1.1. Techniques of matter recovery from sewage sludge

Methods of recovering resources from sewage sludge are shown in Figure 3.8. There are biochemical, thermochemical and mechanical-chemical techniques that result in the recovery of matter and energy from sewage sludge.

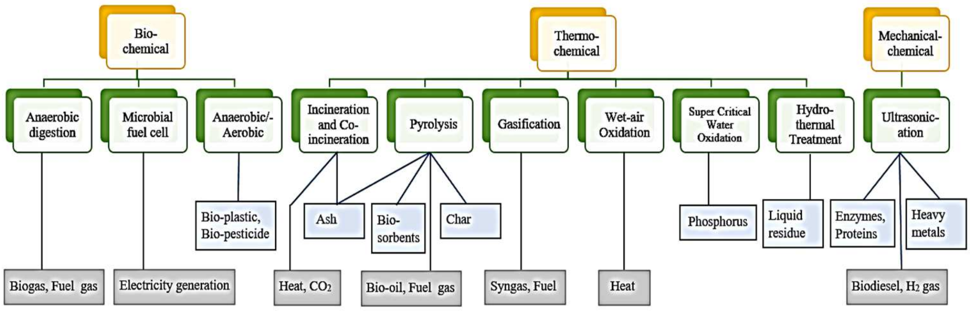

Fig. 3.8. Techniques used to recover matter and energy from sewage sludge (based on Tyagi and Lo, 2013)

3.2.1.1.1. Bioplastic

Bioplastic may be produced through two methods: biosynthesis of micro-organism as an energy storage component such as polyhydroxyalkanoates (PHA); from the chemical formation such as poly(p-phenylene) formic acid diol ester (PTT) and polylactic acid (PLA). The bioplastic is divided into all-bioplastic and part bioplastic. All-bioplastics are the ones which are all derived from biomass materials. The others are part-bioplastics (Liu et al., 2019).

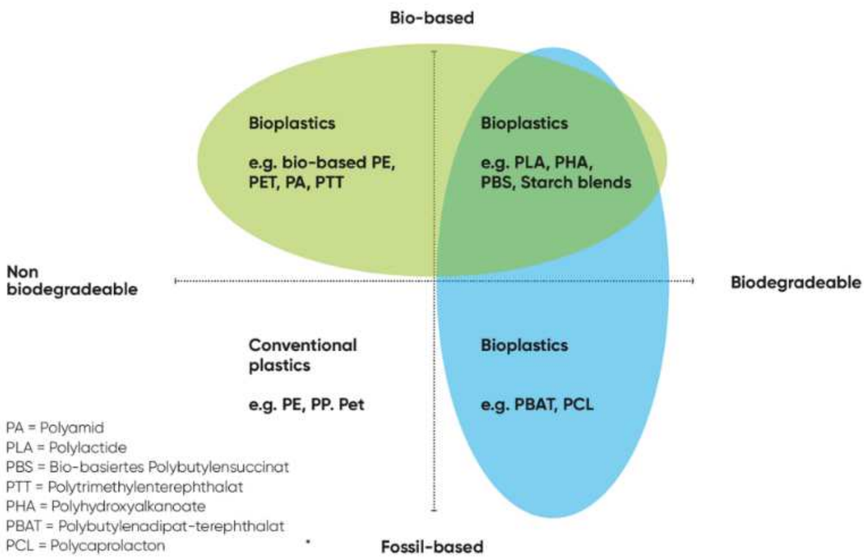

Fig. 3.9. Types of bioplastics (www.european-bioplastics.org/bioplastics)

Part-bioplastics contain starch bioplastic, the bioplastic modified with natural biomaterials like starch and cellulose, in which propylene glycol synthetic monomers are derived from biomass materials, plastic-wood products, plastic--wood products obtained by blending biomass materials with petroleum-based plastics, which is mainly based on PE (polyethylene) and PVC (polyvinylchloride). All-bioplastics include protein plastics, such as soybean fiber, cellulosic, a cellulose derivative obtained by chemical treatment of natural cellulosic materials, algae--based resin (Liu et al., 2019). Bioplastics are also classified into biodegradable bioplastics and non-biodegradable (Fig. 3.9).

3.2.1.1.2. Construction materials

Reusing waste sludge for construction materials can reduce problems with its disposal. A renewable substitute for the depletion of non-renewable resources is also being created. Sewage sludge contains both the organic carbon-containing complexes and inorganic composites (Tyagi and Lo 2013). There are a lot of techniques of thermal oxidation for sewage sludge. The most commonly used methods are: direct incineration of sewage sludge, sewage sludge incineration with household waste, co-incineration of sludge in rotary boilers in the cement industry, and co-incineration of sludge in the energy sector objects. Figure 3.10 presents the technologies of thermal processing of sludge (Smol et al., 2015).

Dried sludge or ash from the incineration plant is used for the production of construction materials (Tyagi and Lo, 2013). They can be used as: a cement production component, a brick production component, as well as active cement additive inorganic binders (Neczaj and Grosser, 2018; Wójcik et al., 2018). Table 3.6 presents the examples of using sewage sludge and other waste in construction materials.

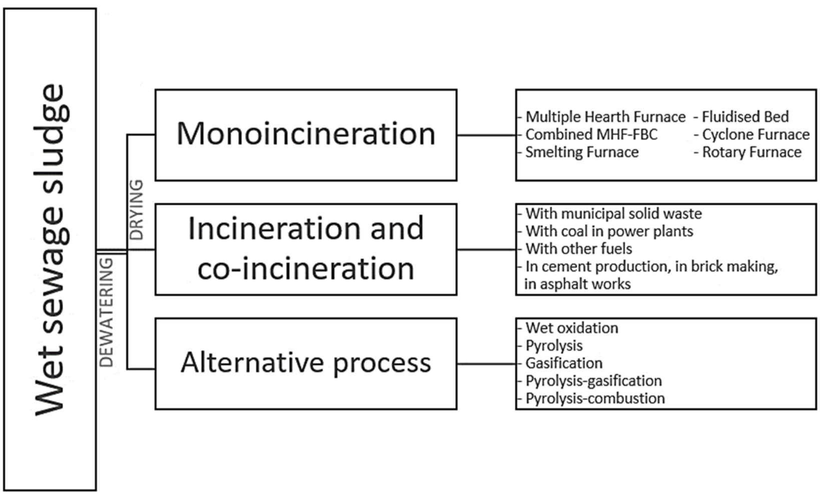

Fig. 3.10. Technologies of thermal processing of sludge (Smol et al., 2015)

Table 3.6

Examples of using sewage sludge in construction materials (Wójcik et al., 2018)

No.	Construction materials	Used types of waste
1	bricks production	sewage sludge, clay, shale
2	bricks production	sewage sludge ash
3	bricks production	sewage sludge
4	pavement base layers	sewage sludge, cement, lime, emulsion
5	as a mineral filter in asphaltic paving mixtures	sewage sludge
6	additive in the production of cement	dried sewage sludge
7	road's embankments	sewage sludge
8	synthetic aggregates	sintered sewage sludge, sewage sludge ash
9	as a component of road base layers	sewage sludge
10	production of cement like material	dewatered sewage sludge, limestone powder
11	production of ceramics materials	sewage sludge ash
12	production of lightweight clay ceramic	sewage sludge
13	production of glass-ceramic material	sewage sludge ash
14	an additive in the production of cement	waste paper sludge ash
15	as an additive in the production of cement material	sewage sludge ash

Methods of using waste sludge as construction materials are not economically viable due to the higher production cost. Therefore, the commercialization of sewage sludge-based construction material are the major challenges in sewage sludge management (Tyagi and Lo, 2013).

3.2.1.1.3. Heavy metals

Heavy metals are the main reason for restricting the use of sludge for land applications due to the possibility of soil and groundwater contamination. The main source of heavy metals in sewage sludge are industrial wastewater and surface runoff. The total content varies within wide limits (from 0.5 to 2% of dry sludge). Taking into account the amount of a single element, it can be ordered as follows: Zn>Cu>Cr>Ni>Pb>Cd or Zn>Cr>Pb>Cu>Ni>Cd (Fijałkowski et al., 2017). The methods applied for the remediation of waste materials include (Tyagi and Lo, 2013): a) thermal treatment using microwave: pyrolysis of sewage sludge and microwave assisted extraction and digestion; b) ultrasonication assisted acid leaching process.

Sonication is the act of applying sound energy to agitate particles in a sample for various purposes. Ultrasonic frequencies (>20 kHz) are usually used, leading to the process also known as ultrasonication or ultra-sonication. The main physical parameters that play vital roles in ultrasound process include power, frequency and amplitude (Wen et al., 2018). The technique of ultrasonication-assisted acid leaching has been implemented at an industrial scale in a heavy metal recovery plant in Huizhou city, China. The pilot-scale installation has treated 5800 Mg of waste sludge from a printed circuit board. From the installation 1000 Mg of 98% copper sulphate and 3500 Mg of 20% ferric chloride were produced. The copper sulphate was sold in the market and ferric chloride was reused in local printed circuit board manufacturing industries. No second pollution was generated by this method (Tyagi and Lo, 2013).

3.2.1.1.4. Phosphorus

Phosphorus is an essential nutrient in the agricultural sector used for the production of fertilizers and feed. There are more than 30 kinds of technologies of phosphorus recovery from sewage sludge and new ones are constantly being created. Sewage sludge contains the second greatest amounts of phosphorus. The only organic waste containing more phosphorus is bone meal, but on a global scale it is produced in much smaller quantities than sludge (Cieślik and Konieczka, 2017).

Phosphorus compounds from sewage sludge can be precipitated directly from sludge or from ashes and slag generated in the processes of their thermal processing (Neczaj and Grosser, 2018). The basic processes of precipitation of phosphorus compounds consist in a hydrolysis reaction with sulphuric acid. Then, phosphorus compounds are mainly precipitated in the form of iron orthophosphates, which show significantly worse fertilizing properties (low water solubility) than calcium phosphates (Eagle et al., 2016).

The popular method of phosphorus recovery from sewage sludge is the thermal hydrolysis of sewage sludge in a sulphuric acid environment, described and used by Kemira in the KREPRO (*Kemwater Recycling PROcess*) process (Fig. 3.11).

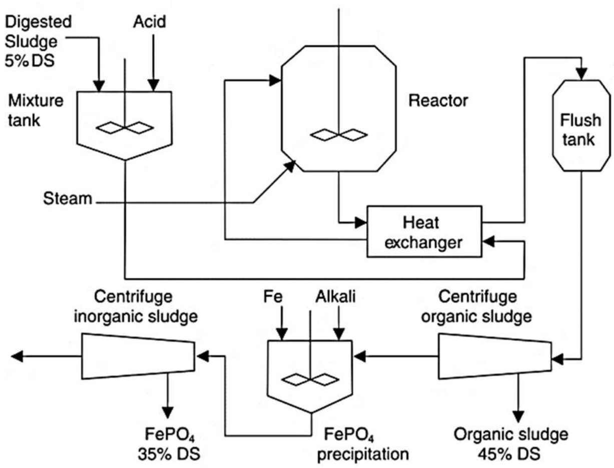

Fig. 3.11. Thermal hydrolysis of sewage sludge in a sulphuric acid environment (KREPRO process) (Hansen et al., 2000)

In the KREPRO process, the sewage sludge (thickened to 5-7% of dry mass) is mixed with sulphuric acid to 1-3 pH. Acidified suspension is heated in the autoclave to the temperature of 140°C. As a result, about 40% of organic matter are hydrolysed into easily biodegradable liquid form. The solution after centrifugation is directed into the reactor, where the pH is raised to the range needed for orthophosphate precipitation in the form of $FePO_4$. The CAMBI/KREPRO process is a development of the KREPRO process. The process can hydrolyse sludge with a dry matter content of 20%. The excess of Fe^{3+} ions necessary for orthophosphates precipitation can be turned back into the neutralizer or used as a coagulant in the wastewater treatment process (Poluszyńska and Ślęzak, 2015; Wzorek and Gorazda, 2007).

The BioCon process involves the recovery of phosphorus compounds from ash resulting from the combustion of sewage sludge at 850°C (Fig. 3.12). The ash is mixed with sulphuric acid to pH ~1. The process uses ion exchange to purify orthophosphoric acid from metal ions. In addition, an absorbable form of phosphates in the form of calcium phosphates is obtained. Table 3.7 presents comparison of Cambi/KRERPO and BioCon systems.

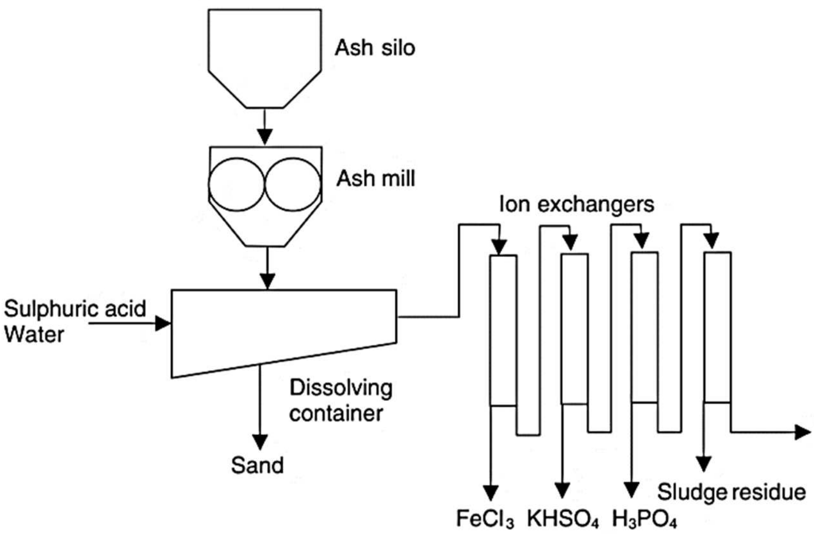

Fig. 3.12. Flow diagram of BioCon process (Svensson, 2000)

The SEABORNE process, developed by Seaborne Environmental Research Laboratory GmbH, involves sulphuric acid hydrolysis of a mixture of sewage sludge and ashes from the thermal transformation of sewage sludge. Hydrolysis of sewage sludge and dissolution of minerals, including phosphorus compounds are proceeded in the reactor. After centrifugation, the organic fraction is incinerated and struvite is formed from the solution (Fig. 3.13) (Poluszyńska and Ślęzak, 2015; Wzorek and Gorazda, 2007).

Table 3.7

Comparison of Cambi/KRERPO and BioCon systems (Levlin et al., 2002)

Process function	Cambi/KREPRO	BioCon
Removal of organic material	Thermal hydrolysis of dewatered sludge, incineration of rest sludge, biological oxidation of soluble hydrolysed organic material	Heat drying and incineration of dewatered sludge
Dissolution of phosphorus and metals	Use of sulphuric acid treatment of dewatered sludge	Use of sulphuric acid treatment of ashes from incineration
Phosphorus product and recovery technology	Ferric phosphate obtained in chemical precipitation	Phosphoric acid obtained by use of ion exchange
Transfer of heavy metals in a small stream	Precipitation as metal sulphides	Concentration by use of ion exchange
Recovery of precipitation agents	Partial recovery by solution by acids (part of the precipitation agent is used to produce ferric phosphate)	Concentrated and recovered by use of ion exchange

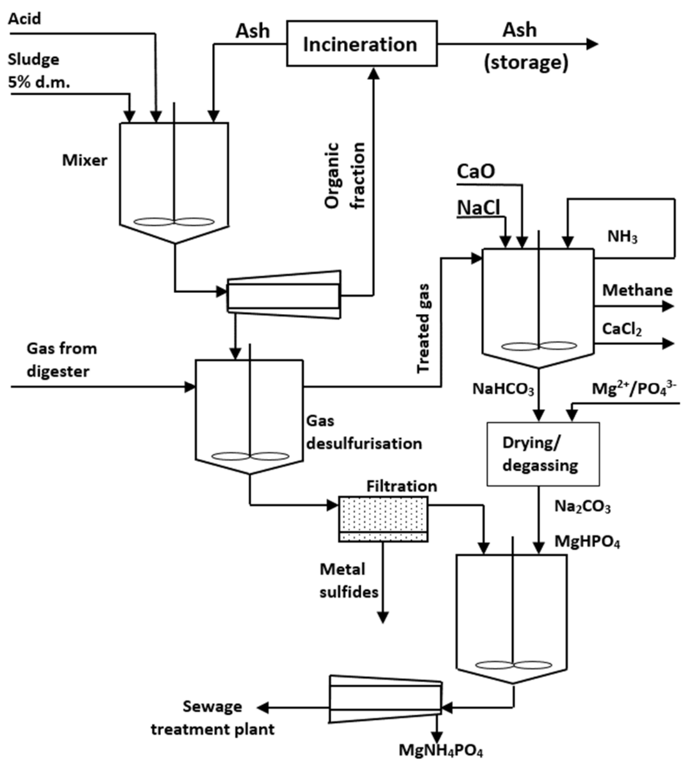

Fig. 3.13. Flow diagram of SEABORNE process (Wzorek and Gorazda, 2007)

The SEPHOS process involves mixing ash containing aluminium ions with sulphuric acid at pH < 1.5 (Fig. 3.14). Then, after separating the suspension, the pH of the solution is raised to 3.5 by addition of sodium hydroxide. $AlPO_4$ and heavy metals are precipitated. Increasing the pH to 12-14 dissolves aluminium phosphate and separates the remaining metals. Phosphates in the form of easily absorbable calcium phosphates are precipitated by the addition of CaO (Poluszyńska and Ślęzak, 2015).

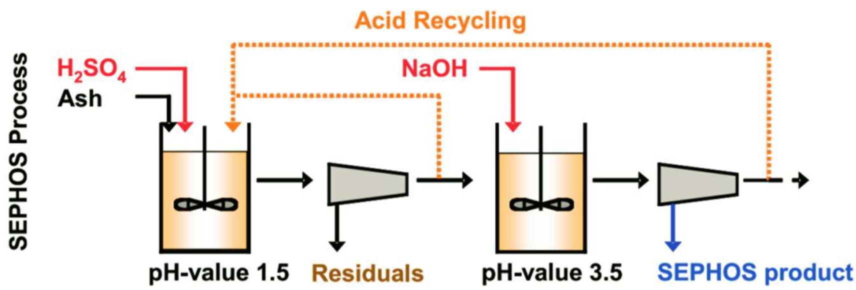

Fig. 3.14. Flow diagram of SEPHOS process (Shaum et al., 2005)

In the Aqua Reci process, the supercritical water oxidation (SCWO) is used to decompose organic contaminants (Fig. 3.15). The oxidation is followed by a chemical process in order to recover components in the inorganic residual ash, like phosphates and coagulants.

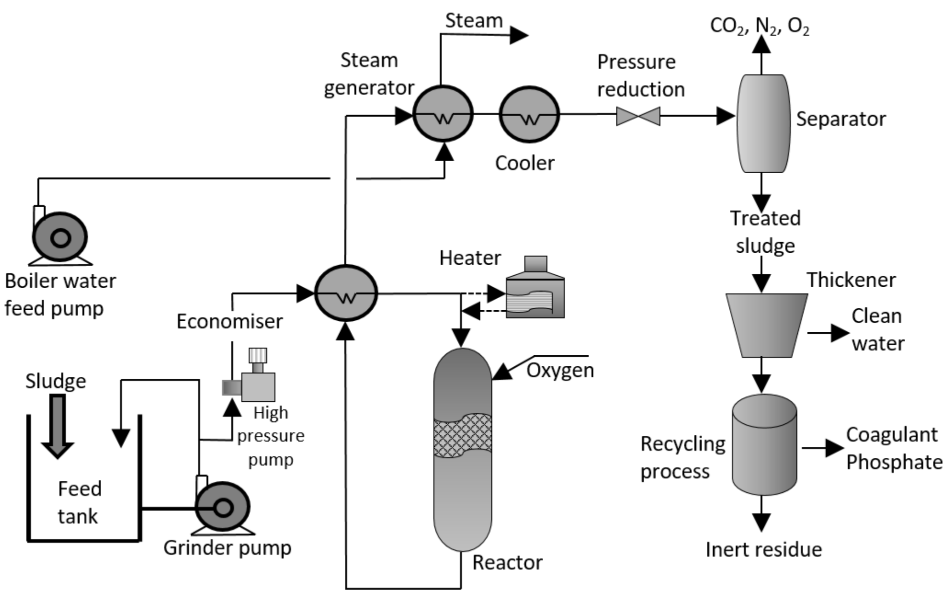

Fig. 3.15. Flow diagram of Aqua Reci process (Stendahl and Jäfverström, 2004)

Table 3.8

Advantages and disadvantages of techniques of phosphorus recovery from sewage sludge (based on Cieślik and Konieczka, 2017)

Techniques of phosphorus recovery	Unit processes of phosphorus recovery	Advantages	Disadvantages
Direct use of sewage sludge in agriculture	– Composting and stabilization in ponds – Stabilization using earthworms – Drying and pellet production	– Low investment costs, – Possibility of managing all sludge in case of small amounts of excess sludge – Low energy expenditure and a reduction in concentrations of heavy metals (in case of earthworm stabilization) – Processes are cost--efficient even with small amounts of excess sludge	– Long stabilization time if low-temperature processes are used – Possibility of contamination of the environment with a variety of organic pollutants, parasites and pathogens, – Applications limited to fertilizers and soil remediation – Methods based on soil remediation not recommended by the European Union
Recovery from sewage sludge and leachates	– Precipitation of phosphorus in form of struvite, hydroxyapatite	– Low probability of releasing heavy metals – Slow phosphorus release – Possibility of solving the problem of clogging pipes	– High investment costs, – Possibility of contamination of the environment with a variety of organic pollutants, parasites and pathogens – Applications limited to fertilizers – Incomplete phosphorus recovery – Incomplete management of sewage sludge – Incurred sewage sludge management costs are not fully recovered
Recovery form ashes after sewage sludge incineration	– Incineration, – Acidic extraction – Thermochemical treatment – Cementing	– Partial refund of costs, – Considerable savings associated with waste disposal – Complete management of sewage sludge, – High phosphorus recovery efficiency – Possibility of simultaneous treatment of some heavy metals – Possibility of energy recovery during incineration process – Less odours	– The highest investment costs – Possibility of contamination of the environment with heavy metals and some organic pollutants – Problems with obtaining high strength of the produced building materials – Processes are cost-efficient only with large amounts of excess sludge

The Aqua Reci process is the wet oxidation process for sewage sludge, which is run in the supercritical range of water (p > 221 bar, T > 374°C). SCWO leads to complete destruction: organic carbon is converted into carbon dioxide, organic and inorganic nitrogen into nitrogen gas (N_2), halogenated organics and inorganics into corresponding acids, and sulphonated organics and inorganics into sulphuric acid. Metals are oxidised to their highest valency and phosphorus into P_2O_5. Phosphorus can be extracted with caustic and separated from metal oxides due to their insolubility in alkaline conditions (Stendahl and Jäfverström, 2004).

The cost of phosphorus recovery from waste sludge is high compared to the cost of mined phosphate rock. Therefore, this is the main obstacle hindering the scale-up of the processes being investigated. Most of the methods are energy-intensive, and the research is carried out on a laboratory or pilot scale (Tyagi and Lo, 2013). Table 3.8 presents advantages and disadvantages of the techniques of phosphorus recovery.

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Chapter 4

Review of technologies for the recovery of energy from sewage sludge/biowaste

Ewa OKONIEWSKA

4.1. INTRODUCTION

The excessive production of sludge as a by-product of wastewater treatment is a significant problem worldwide. According to the Polish Statistical Office (2021), in Poland, in 2020, 568.86 thousand Mg total solids (TS) of municipal sewage sludge was produced, which is 60% more than in 2000. Almost 11 million Mg TS of sludge is produced annually in Europe. Furthermore, it is expected that the amount of produced sewage sludge will be increased due to the rising number of wastewater treatment plants, improving the treatment process as well as the increasing the number of inhabitants connected to the sewage network (Rosiek, 2020).

In many countries, including Poland, there is a ban on storing sewage sludge, which in turn makes it necessary to look for new methods of its disposal (Wójcik, 2018; Ding et al., 2021). In line with the strategy adopted in the European Union to reduce waste disposal by 50% by 2050, measures are promoted to enable the conversion of sludge into fuel or product with potential commercial use (Fytili, 2008; Kacprzak and Kupich, 2021; Lundin, 2004).

Biowaste is also becoming a problem. Biowaste is a broader concept than just green waste. In addition to green waste, this concept includes food waste and kitchen waste from households, catering, mass caterers, retail trade units as well as waste from food processing plants. 64% of the energy produced from biomass comes from wood and waste, 24% from municipal solid waste, 5% from agricultural waste, and 7% from municipal waste from landfills (Ordza and Rybska, 2014). At present, biogas energy ranks 5th in the European Union as an alternative energy source (Polish Statistical Office, 2021).

Biomass is defined as an organic substance that can be of origin plant or animal metabolism, but it can also arise as a result of so-called social metabolism. Biomass is characterized by a low content of element C, with a high oxygen content, while sulfur (and nitrogen) occur in small quantities and have little influence on the pyrolysis process.

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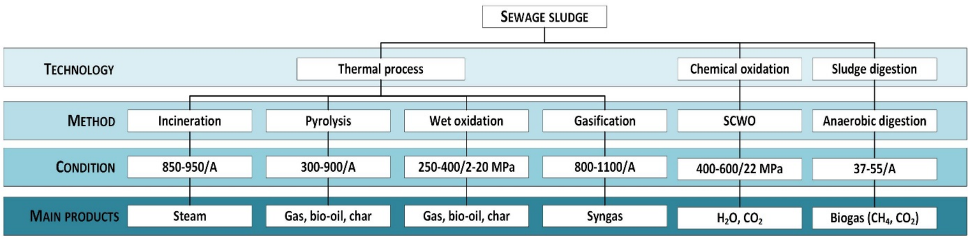
In recent years, an intensive increase in the share of thermal methods in sewage sludge management has been observed, among which mono- and co-incineration have become the most popular. Moreover, energy recovery from sewage sludge is performed using the following technology: pyrolysis, gasification and wet oxidation, and anaerobic digestion (Kacprzak and Kupich, 2021; Przydatek and Wota, 2020). The goal of this chapter is to briefly describe the mentioned methods. The main products and conditions of a process are shown in Figure 4.1.



SCWO – Supercritical water oxidation
Condition of process: temperature/pressure (A – ambient)
Gas – mainly consisting of H_2,CH_4, CO_2 as well as in small concentration contains low molecular weight hydrocarbons
Char – mostly solid carbon and ash (with significantly amount of heavy metals and other inert substances)
Bio-oil – a mixture of oils and/or tar, particularly hydrocarbons, organic acids and carbonyl compounds of high molecular weight phenols, aromatic compounds, aliphatic alcohols, acetic acid and water
Syngas – a mixture consisting primarily of hydrogen, carbon monoxide, and very often small amount of carbon dioxide and methane

Fig. 4.1. Energy recovery methods from sewage sludge (based on Grosser, 2019)

4.2. THERMAL TREATMENT OF SEWAGE SLUDGE

The advantage of management methods that include a stage of thermal stabilization (conducted at temperatures above 700°C) over low-temperature biological and chemical methods is complete mineralization of compounds present in the sediment (Donatello, 2013; Houillon, 2005). Organic substances present in the sewage sludge are oxidized to simple inorganic compounds such as carbon dioxide and water under the influence of high temperatures. Unfortunately, toxic substances such as sulphur nitrogen oxides or carbon monoxide are also produced, so the necessity to remove these substances from the resulting gases must be taken into account. Afterburning and flue gas purification are key elements of the operation in this case for environmental reasons. Thermal processes involve removing the organic part of the sludge, leaving only the ash component for final disposal (Cieślik, 2016; Smol et al., 2020).

An alternative to sludge disposal in the category of thermal sludge use can be pyrolysis, gasification, wet oxidation, and combustion. Sewage sludge is a type of fuel from biomass and its calorific value is similar to that of coal. The main purpose of the thermal treatment of sewage sludge is to use the stored energy in the sludge and, at the same time, minimize the negative impact on the environment. It is well known that sewage sludge contains large amounts of moisture, therefore most of the energy released during thermal processes is used to reduce moisture (Dennis, 2005; Schnell et al., 2020).

4.2.1. INCINERATION OF SEWAGE SLUDGE

Sewage sludge can be burned together with, among other things, solid municipal sludge or fossil fuels. The selection criteria for sludge combustion in different types of boilers depends on the composition of the mixture and the calorific value of different fuels. Fermented, dehydrated and, if possible, dried sludge is used for combustion. Several technologies of thermal treatment of sewage sludge have been developed on the market over the last decades. Mono- and co-combustion of sewage sludge is perhaps the most established technology, with mono-combustion dominating even further. Fireplace stoves and fluidized bed stoves are the most popular, especially since the latter are increasingly used (Smol, 2020; Werther, 1999). The difference between these two types of stoves is that multi-fuel stoves usually burn mechanically dehydrated (wet) sludge, while fluidised bed stoves can burn both wet and semi-dry sludge with a dry matter content of 41-65% mass.

The combustion process produces flue gases, slag, ash, and substances from flue gas cleaning. They are not indifferent to the environment; therefore a comprehensively designed incineration plant must take into account ways of neutralising them. Flue gas purification is the most expensive operation. It is estimated that it can represent 30-35% of the overall costs. During combustion, or even when using gasification, the presence of heavy metals remains a problem. Emissions of mercury, dioxins, and furans are controlled.

The most common method that allows for the energetic use of sewage sludge is incineration in professional incinerators or co-incineration in industrial equipment such as boilers or rotary kilns (Wilk et al., 2008). This process requires the appropriate processing of sewage sludge (removal of moisture to less than 20%).

Combustion may be realized into fluidized bed ovens. This process causes the organic components of the sludge to start burning and it transforms into fine particles, which are carried out form chimney with the steam and flue gas mixture. The requirements for the complete elimination of organic substances are practically fulfilled at 100%. An alternative to fluidised bed boilers is the combustion of sewage sludge in a cement rotary kiln. The rotary kiln is inclined at a certain angle and rotates at a slight speed. By rotation, the sewage sludge is mixed, passing through the following zones: drying, degassing, spacialisation, and chilling the ashes. The combustion process takes place at a temperature of 800-1000°C. The advantage of combustion in rotary furnaces is thermal stability in relation to load and quantity changes of the material being fed. The disadvantage is the sensitivity to change of heat load and unfavourable energy balance. High gas flow temperatures (> 2000°C) and the material to be burned (approx. 1450°C), turbulence and the relatively long amount of time (7-10 s) of gas and material flow in the high temperature zone (> 1200°C) cause the combustion process to the sewage sludge in the rotary kiln meets all standards. Conditions the combustion of organic combustible substances introduced into the furnace results in almost complete decomposition and combustion. The advantage of a cement rotary kiln over a sludge incinerator or other equipment is its waste free disposal. The resulting spa product is ash and is completely absorbed and permanently bound in the clinker without posing a risk to the environment (Duda et al., 2014; Gao et al., 2020; Środa et al., 2012; 2013).

Combustion requires further drying of dehydrated sludge to reduce moisture content to < 50% by weight before entering the reactor, as shown in Figure 4.2.

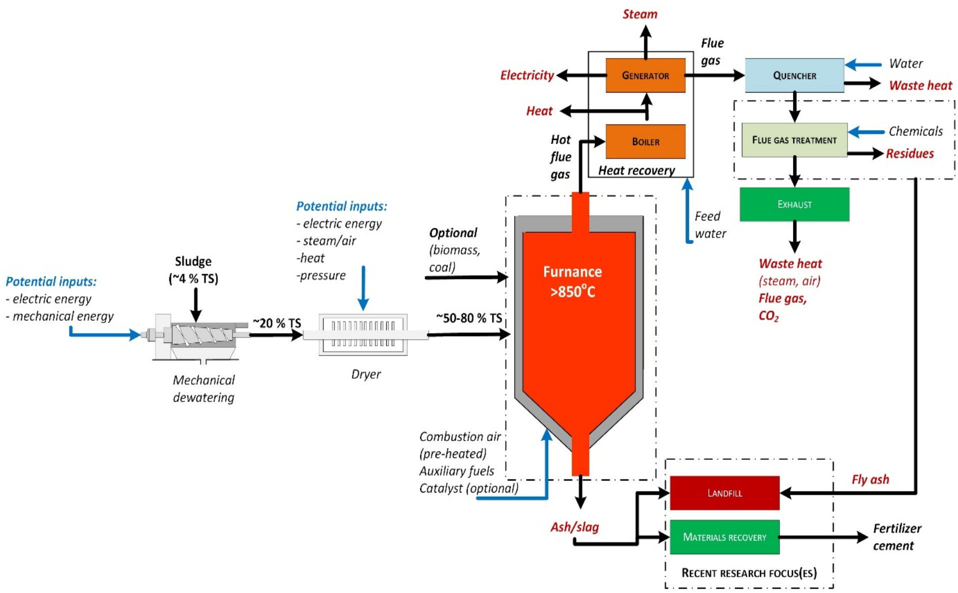


Fig. 4.2. Sludge combustion scheme (based on Oladejo et al., 2019)

Sewage sludge has a similar calorific value to coal sludge, lignite, or firewood, however, coal sludge has a higher ash content. The amount of elemental coal is similar to that of coal sludge and firewood, while the amount of hydrogen is similar to that of conventional fuels. Sewage sludge has large amounts of elemental sulphur in comparison with other fuels. Only lignite and waste coal sludge contain larger amounts of this element (Cano, 2015; Carrere, 2016; Zhang, 2019; Zhen, 2017).

Dry mass to be burned in highly dehydrated sludge is usually between 18 and 35%. Although the calorific value of sewage sludge is at a horizontal level of lignite, its combustion temperature (700-900°C) is much lower than that of coal (1500-1700°C), due to the water content of the sludge. After drying, sludge can contain about 70% of water by weight. Part of the energy obtained from the combustion of sludge is therefore used for its evaporation. For this reason, the energy value of dried sludge (12-20 MJ/kg) is usually lower than that of hard coal (14.6-26.7 MJ/kg) and comparable to that of lignite (11.7-15.8 MJ/kg). Reduction of the degree of hydration of dried sludge from 77 to 6.5% by weight requires almost 8 MJ/kg. It is possible to reduce the energy consumption by even half if the initial product instead of 77% contains 65% of H_2O by weight. The process of thermal sludge disposal, according to current regulations, should take place at a temperature of at least 850°C (1100°C if the sludge contains significant amounts of both chlorine and its compounds). In the case of a low degree of sludge dehydration, the thermal treatment process may not be carried out auto-thermally, making it necessary to add other flammable media to the process (methane or fuel oil) (Samolada, 2014).

4.2.2. PYROLYSIS

Pyrolysis (gr. pyro, πῦρ – fire, and lysis λύσις – decay) means the decomposition of molecules of a chemical compound under the influence of elevated temperatures without the presence of oxygen or another oxidizing agent. Pyrolytic process means an endothermic process decomposition of organic matter in an anaerobic environment. The process is carried out from a temperature range of 250-900°C. Wandrasz (1998) defined the pyrolysis process carried out in a tempera-hole up to 600°C as extrusion and above 600°C as degassing. Both organic (e.g. coal, biomass, waste) and inorganic (ceramic raw materials) materials can undergo the pyrolysis process (Kramer et al., 2009). Due to the applied process temperature in the literature, pyrolysis can be divided into pyrolysis (Rosik-Dulewska, 2010):

- low-temperature pyrolysis – with low-temperature pyrolysis, the waste undergoes conversion at 450-500°C,
- at high temperature – with high-temperature pyrolysis, the waste undergoes a conversion at 700-800°C.

Under these temperature conditions and in the absence of oxygen from the air, organic matter is converted into process products such as pyrolysis gases, pyrolysis oils and coke, and in the case of biomass pyrolysis, a fraction called biocarbon. The production of biochar from different types of biomass in the pyrolysis process makes it possible to liquid and gaseous fuels for energy production, and the resulting biochar is a solid, renewable fuel used in the power industry (Fig. 4.3) (Wang et al., 2020).

The composition and quantity of these products depend on a number of factors, including: the type of waste, its physico-chemical properties, the range of temperatures used, and the dwell time in the reactor. The pyrolytic gas consists mainly of hydrogen, methane, carbon monoxide, carbon dioxide, steam, higher aliphatic hydrocarbons and other gaseous compounds such as hydrogen sulphide, hydrogen chloride, hydrogen fluoride, ammonia. These gases may also contain dust of relatively high content of heavy metals. Pyrolytic oil is a mixture of oils, tar, water and organic components. On the other hand, pyrolytic coke contains elemental carbon and mineral substances with a high concentration of heavy metals. Mutual proportions of the individual phases are a function of temperature. The largest share of products is usually the liquid phase, which is about 60% (Huang et al., 2020).

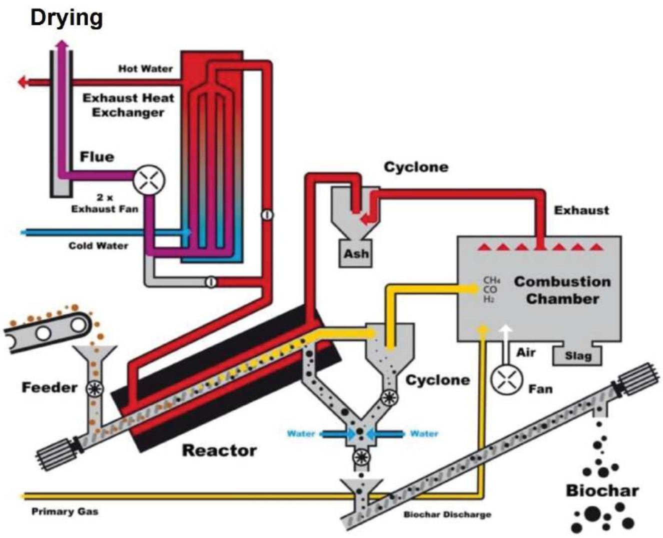


Fig. 4.3. Sewage sludge pyrolysis installation (based on Tsybina and Wuensh, 2018)

Pyrolysis is a complex process of breaking down chemical compounds into smaller molecules under the influence of heat supplied from outside (Kardaś et al., 2004). Most reactions during pyrolysis are endothermic.

Pyrolysis produces products that form three fractions:

1) solid product (biochar, carbonisate, pyrolytic coke,
2) the liquid fraction (tar and/or biooil),
3) a gaseous fraction, being a mixture of CO_2, CO, H_2, and hydrocarbons, mainly methane.

The waste is therefore anaerobically transformed by heating it to a maximum of 1000°C. The aim of the process is to reduce the amount of waste, make it sanitary and produce so-called pyrolytic gas, which can then be conventionally incinerated with energy recovery. Its calorific value can range from 8 to 30 MJ/m^3.

Each of the pyrolysis products can be used to produce heat and electricity (the calorific value of a pyrolytic coke is up to about 30 MJ/kg and of the liquid fraction 16-19 MJ/kg, while the gas ranges from 5 to 36 MJ/kg). In addition, the liquid fraction can be used for fuel production, while biocarbon can be used as an adsorbent to remove organic and inorganic pollutants, e.g. from wastewater, or as a soil conditioner, where it can also be used for carbon sequestration (Raheem et al., 2018).

Due to the process conditions used, such as temperature, heating rate, degree of grinding, and end-temperature heating time, we distinguish between several types of pyrolysis, as shown in Table 4.1.

Table 4.1

Parameters of different types of pyrolysis (based on Tripathi et al., 2016)

Process conditions	Slow (conventional)	Fast	Instantaneous	Intermediate	Vacuum	Hydro-pyrolysis
Temperature, °C	550-950	850-1250	900-1200	500-650	300-600	350-600
Warm-up rate, °C/s	0.1-1.0	10-20	> 1000	1.0-10	0.1-1.0	10-300
Response time, s	300-550	0.5-10	< 1	0.5-20	0.001-1.0	> 15
Pressure, MPa	0.1	0.1	0.1	0.1	0.01-0.02	5-20
Particle size, mm	5-50	< 1	< 0.5	1-5	–	–

4.2.3. WET OXIDATION

Wet oxidation has been in commercial use for about 60 years. It is mainly used for wastewater treatment. It is often referred to as Zimpro (from ZIMmerman PROcess) to Fred J. Zimmermann, who commercialised it in the mid-20th century. A block diagram of the Siemens Zimpro® wet air oxidation system is shown in Figure 4.4.

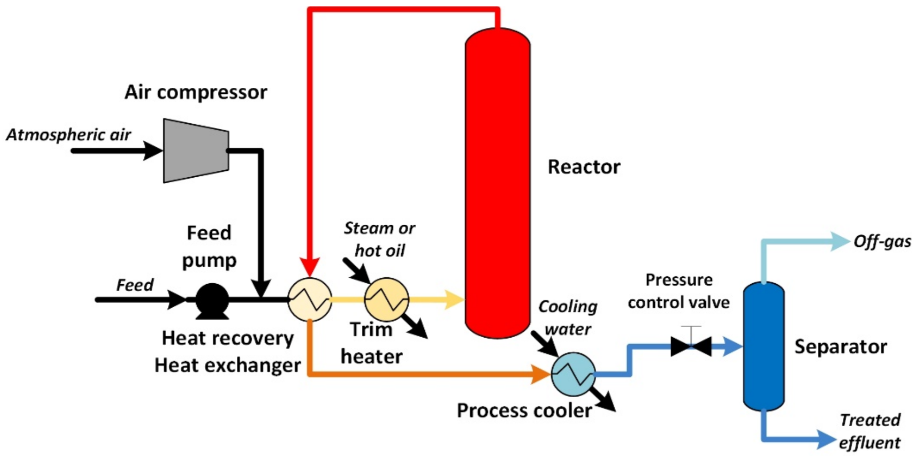


Fig. 4.4. The block diagram of the Siemens Zimpro® wet air oxidation system (https://press.siemens.com/global/en/feature/siemens-successfully-commissions-zimpror-wet-air-oxidation-unit-olefins-plant)

Wet sludge oxidation is classified as a thermal process. It takes place in the aqueous phase at 150-330°C and at a pressure of 1-22 MPa in the presence of homogeneous and/or heterogeneous catalysts. The optimum concentration of the solid phase for the wet oxidation process is 5-10% d.m., which makes it possible to dispose of sewage sludge without mechanical dewatering. High temperature is necessary to prevent boiling at temperatures required for the process. During the process, the content of organic substances in the sewage sludge is thermally decomposed, hydrolysed, oxidized, and converted into carbon dioxide, water and nitrogen (Malhotra and Garg, 2021). The whole process takes place under two separate regimes:

1) the process under subcritical conditions, below 374°C and at 10 MPa;
2) the process under supercritical conditions below 374°C and at a pressure of 21.8 MPa (Fytili, 2008).

During wet oxidation a number of different reactions take place, which can be recorded as a single equation as follows:

$$C_aH_bO_cN_dP_eS_fX_g + yO_2 \rightarrow CO_2 + H_2O + NH_4 + PO_4^{3-} + SO_4^{2-} + XO +$$
$$+ \text{fatty acids} + \text{energy}$$

Then carbon is converted to carbon dioxide, organic nitrogen to ammonia or free nitrogen, while organic chlorides and sulphides are converted to inorganic chlorides and sulphates. Water and small amounts of low-molecular organic acids are also produced, which are responsible for the decrease in organic carbon oxidation rate. During the process, no pollutants such as NO_x, SO_2, HCl, HF, dioxins, furans or fly ash are emitted to the atmosphere, which takes place during the thermal disposal of sediments. The final product is, inter alia, post-reaction gases, in which the composition, calculated as dry gas, is contained: CO_2 (80% by volume), CO (3% by volume), O_2 (16% by volume) and N_2 (1% by volume) (Bień et al., 2011; Malhotra and Garg, 2021).

4.2.4. GASIFICATION

Gasification (called quasi-combustion or quasi-pyrolysis) is the process of converting solid fuel into gaseous fuel by interacting with oxygen (or air) and steam on the solid fuel contained in a generator. The gasification process takes place at a temperature of between 700 and 1400°C and involves a number of chemical and thermochemical transformations taking place during the contact of the material to be converted with the gasification agent (for example, oxygen, air, steam, hydrogen, hydrocarbons and carbon dioxide are used for this purpose) (Molino et al., 2018; Shahabuddin et al., 2020).

An example of the gas composition from sewage sludge gasification is shown in Table 4.2.

Table 4.2

Typical content of combustible components in sewage sludge gasification gas (Werle et al., 2009)

Component	% vol.
Carbon monoxide	6.28-10.77
Hydrogen	8.89-11.17
Methane	1.26-2.09
Ethane	0.15-0.27
Acetylene	0.62-0.95

The calorific value of the gasification gas varies around 4 MJ/m^3. Sewage sludge gasification leads to a combustible gas which can be used in the electricity generation process or as a fuel to assist in the sludge drying process (Marrero et al., 2004).

During the high-temperature gasification process, hydrogen is produced (Mathieu et al., 2002). Solid and fluidised bed reactors are used in the sewage sludge gasification process. It is recognised that fluidised bed boiler technologies should be designed for systems with relatively large capacities above 10 MW. More and more often, in-house boiler plants are being built at sewage treatment plants, which are equipped with sewage sludge-fired boilers (Fig. 4.5).

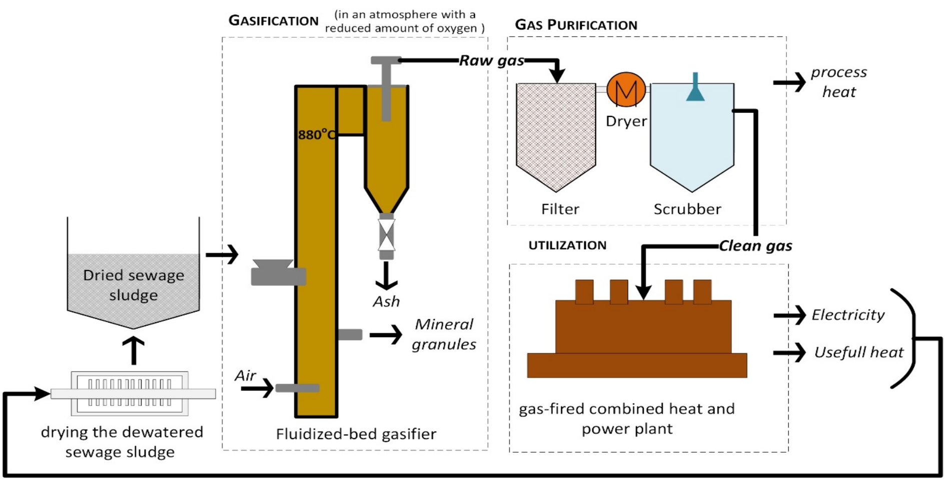


Fig. 4.5. Sewage sludge gasification (based on Urbański, http://www.forum-wodociagi.pl/artykuly/98-zagospodarowanie-osadu-sciekowego-utylizacja-zamiast-skladowania.html)

On 18 October 2002, a pilot installation based on the technology of incineration of sewage sludge by the gasification method was launched on the premises of the sewage treatment plant in Balingen (Baden-Württemberg). The plant in Balingen is operated on an industrial scale and processes approximately 150 kg of sludge with 85% total solid per hour. Converted to dehydrated sludge with 25% total solid, the plant has a gasification capacity of approx. 1,000 Mg of total solid per year of sludge per day (Fig. 4.5).

4.3. ANAEROBIC DIGESTION

Another method for energy recovery from sewage sludge, which fits well in renewable energy policies in the EU, is anaerobic digestion, also known as anaerobic or anaerobic digestion. It is one of the oldest biochemical transformations on Earth, occurring naturally thanks to the activity of appropriate microorganisms

(Krupa, 2015). Anaerobic digestion comprises four stages: 1) hydrolysis, 2) acidogenesis, 3) acetogenesis, 4) methanogenesis.

In the technological sense, this process has been known to man for about 150 years.

Different sources of production lead to different specific compositions of the biogas and is shown in Table 4.3.

Table 4.3

Examples of biogas composition (Speight, 2018)

Constituents	Household waste	Wastewater treatment plant sludge (WWTP)	Agricultural waste
Methane, CH_4, % v/v	50-60	60-75	60-75
Carbon dioxide, CO_2, % v/v	38-34	33-19	33-19
Nitrogen, N_2, % v/v	5-0	1-0	1-0
Oxygen, O_2, % v/v	1-0	<0.5	<0.5
Water, H_2O, % v/v	6 (40°C)	6 (40°C)	6 (40°C)
Hydrogen sulfide, H_2S, mg m^{-3}	100-900	1000-4000	3000-10 000
Ammonia, NH_3, mg m^{-3}	–	–	50-100

The last decade has seen significant development towards the use of energy crops, industrial and municipal waste for anaerobic digestion. Biogas is produced in anaerobic digestion plants, from wastewater treatment and landfill recovery. In Europe, biogas is mainly produced from anaerobic digestion in anaerobic chambers using agricultural waste, manure and energy crops (Table 4.4).

Table 4.4

Biomethane yield from selected feedstocks (Scarlat, 2018)

Substrate	DM	VS	Methane yield	Methane yield
	%	% of TS	CH_4/kg VS	1 CH_4/kg fresh feedstock
Pig slurry	3-8	70-80	250-350	6-22
Cattle slurry	6-12	70-85	200-250	8-25
Poultry manure	10-30	70-80	300-350	21-84
Maize sillage	30-40	90-95	250-450	68-170
Grass	20-30	90-95	300-450	55-128
Alfalfa	20-25	90-95	300-500	27-118
Potatoes	20-30	90-95	280-400	54-128
Sugar beet	15-20	90-95	230-380	31-72
Straw	85-90	80-90	200-250	136-202
Vegetable waste	85-90	80-90	200-251	136-203
Organic waste	10-40	75-90	350-450	26-180
Slaughterhouse residues	35	90-95	550-650	173-216
Sewage sludge	5-10	75	300-400	11-30

TS – Total Solids, VS – Volatile Solids

Depending on the location of biogas production, we distinguish the following sources of mentioned gas:
1 – landfills;
2 – wastewater treatment plants;
3 – other sources (agricultural).

These compounds together form biogas, which is a high calorific value fuel that ranges from 19 to 25 MJ/m^3. For comparison – the calorific value of natural gas is about 36 MJ/m^3 (Grosser, 2017; Raheem et al., 2018). However, using methods of biogas upgrading consisting of the removal of the abovementioned impurities, it is possible to increase its calorific value to the level of 35.7 MJ/m^3, i.e. to a value similar to that of variable gas.

Biogas resulting from the anaerobic decomposition of bio-waste and manure is a combustible gas and can be used to generate heat and electricity or can be converted to biomethane and used as a transport fuel or fed into the natural gas network (Grzesik, 2006; Kapoor et al., 2020). Such solutions are well known in Sweden as well as in Germany or Austria.

Mulchandani et al. (2016) states that 1 kg of removed sewage sludge can yield between 0.75 and 1.12 m^3 of biogas and per person per day between 0.03 and 0.04 m^3/d. Grzesik (2006) states that from 1 m^3 of liquid animal manure, an average of 20 m^3 of biogas can be obtained, and from 1 m^3 of manure – 30 m^3 of biogas, with an energy value of about 23 MJ/m^3.

The use of biogas combines the energy and environmental effect because it eliminates environmental pollution while producing energy. During the combustion of biogas less harmful nitrogen oxides are produced than in the case of fossil fuels. An important benefit is also the reduction of carbon dioxide and methane emissions into the atmosphere, which in turn contributes to the reduction of the greenhouse effect. Increasingly, investments are being made in numerous methane recovery installations (Thiruselvi et al., 2020).

The best results of biogas production are obtained in biological wastewater treatment plants, which at the same time have a high demand for heat and electricity. Biogas as a result of combustion is converted into electricity or heat in cogeneration systems.

To increase the biodegradability of sludge and simultaneously improve biogas production, it is necessary to condition the sewage sludge before it enters the digesters. These methods can be divided into mechanical, chemical, biological, and thermal, depending on the nature of the conditioning agent. Intensification of the anaerobic digestion process is also possible through co-digestion, i.e. joint fermentation of sewage sludge with other organic waste, e.g. with food waste, selectively collected organic fraction of municipal waste, waste from sugar factories or waste from animal slaughterhouses (Grosser, 2017).

Biogas can be used: for steam or heat generation, as fuel gas, for hydrogen production, as a substitute for natural gas, for SOFC fuel cells, and during the synthesis of liquid fuels such as methanol (Kapoor et al., 2020).

In some countries, small scale fermentation plants for the animal, human and livestock waste are used to power central heating furnaces. The largest number of such plants is in China (about 6 million), India (1 million), South Korea, Brazil and Nepal. The Americans specialize in building fermentation plants that process manure from farms with several thousand cows each. The French have mastered the technology of producing biogas from waste from vegetable and industrial plants. In Scandinavian countries, heat from the fermentation of human and animal manure is used to heat their homes. The Danish biogas plants currently produce over 260 GWh of electricity per year. In Germany, there are more than 600 agricultural biogas plants located mainly on individual farms. Between 2009 and 2018, the total number of biogas plants in the European Union increased from 6,227 to 18,202 installations. Germany, Italy, Denmark, Czech Republic, Austria and Greece (Brewer, 2020) dominate the number of biogas plants in their territories. German biogas production currently accounts for about 50% of the production in the European Union (Theuerl, 2019).

The reduction of volume of sludge as well as production renewable energy (biogas) is obtained during fermentation process (anaerobic digestion). The disintegration of sludge directly affects to increase in biodegradability of sewage sludge, which allow on the intensification of the anaerobic stabilization process. Intervention in the anaerobic digestion process by modifying sludge before the stabilisation process affects their final susceptibility to dewatering (Zawieja, 2019).

The share of conditioning methods in the pre-treatment of different substrates is shown in Figure 4.6.

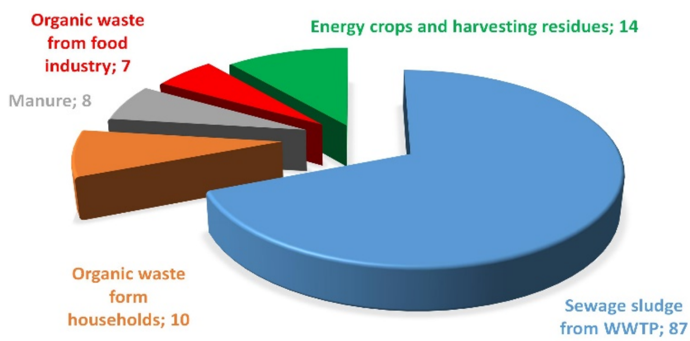


Fig. 4.6. The share of conditioning methods in the pre-treatment of different substrates (Neczaj, 2016)

One of the interesting commercial solutions is the JFE BIGADAN biogas system (Fig. 4.7), that produces methane by fermentation of animal manure from slurry, food waste, sewage sludge, etc. At a temperature of about 37°C, electricity and liquid fertilizer are recovered. The BIGADAN-type anaerobic digestion process is mainly characterized by complete sterilization by heating at a temperature of about 70°C for one hour and recovery of heat from the suspension. The BIGADAN-type biogas system is widely used in Denmark.

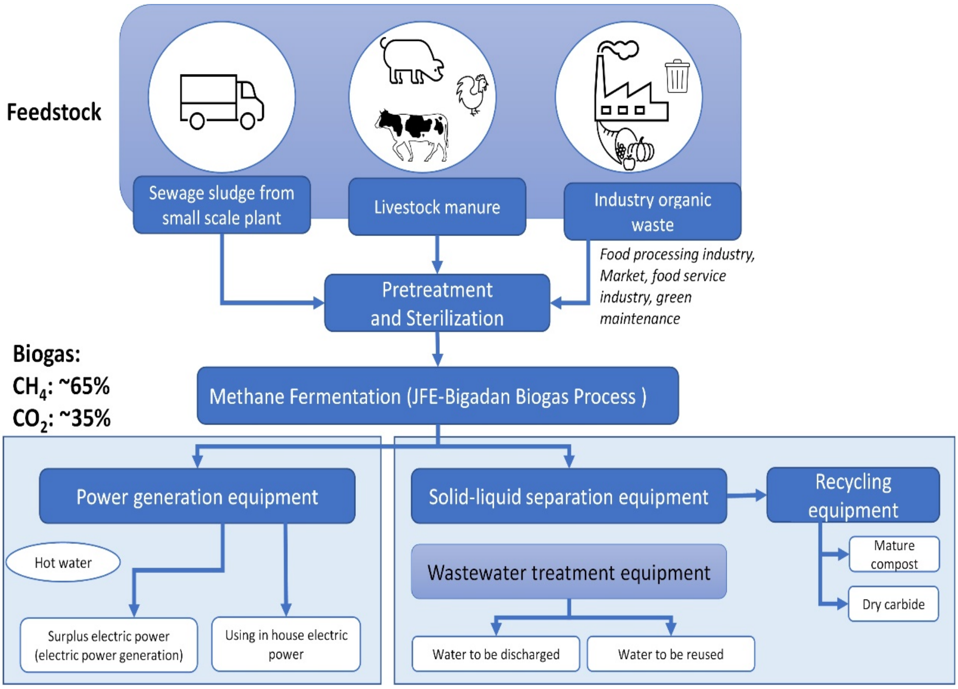


Fig. 4.7. JFE BIGADAN biogas system (https://www.jfeeng.co.jp/en/products/aqua/aqua02.html)

In the process of agricultural biogas production, with less manure use and plant waste, about 50% more of this fuel can be produced than in landfill biogas production.

Data for 2019 indicate that in Poland the number of installations included in the register of agricultural biogas producers is 103 (as of 24 March 2020). (Data on the activity of agricultural biogas producers in 2011-2019, http://bip.kowr.gov.pl/informacje-publiczne/odnawialne-zrodla-energii/biogaz-rolniczy/dane-dotyczace-dzialalnosci wytworcow-biogazu-rolniczego-w-latach-2011-2019).

Locations of functioning agricultural biogas plants in Poland are shown in Figure 4.8. For instance, in Poland, 10 years of experience in co-digestion of sewage sludge and biodegradable, e.g., this process was implemented Wastewater Treatment Plant in Tychy-Urbanowice (Fig. 4.9). Since 2006, it has also been a producer of electricity and heat from renewable sources – an ecological fuel, which is biogas. Surplus electricity and heat were the beginning of the investment in the Tychy Water Park. Figure 4.9 shows a diagram of a biogas installation at the sewage treatment plant in Tychy-Urbanowice.

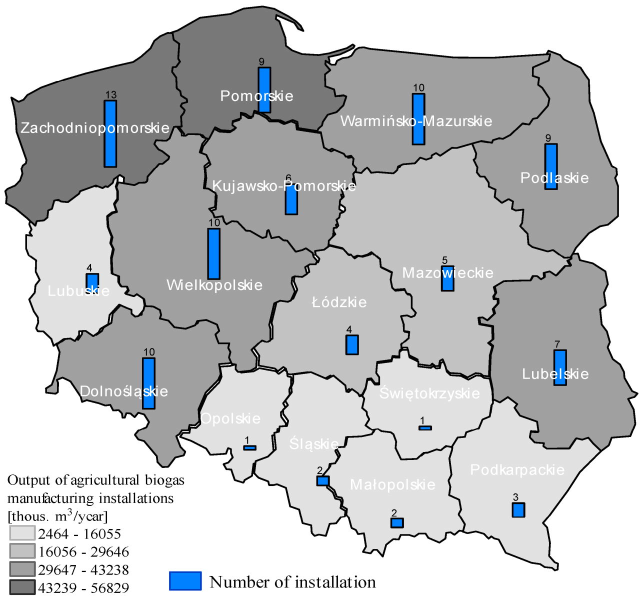


Fig. 4.8. Number and capacity of agricultural biogas plants in Poland (as of 24 January 2019)

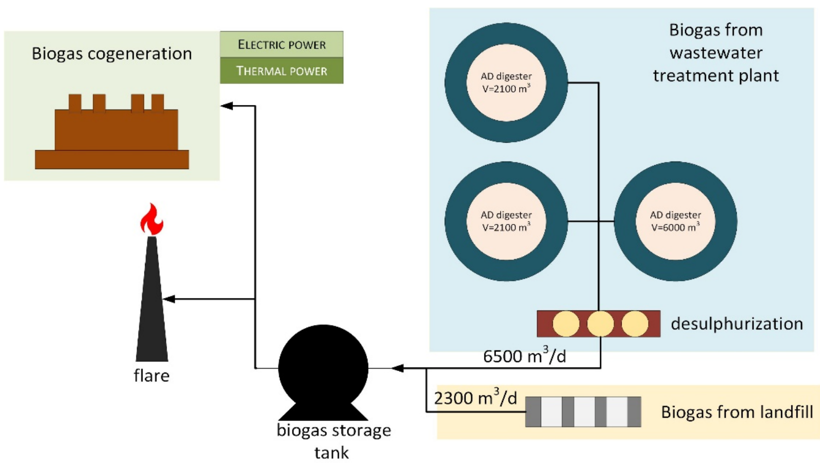


Fig. 4.9. Schematic diagram of a biogas plant at the sewage treatment plant in Tychy-Urbanowice (based on Grzesik, 2006)

The management and disposal of municipal waste is usually landfill, which is a kind of bioreactor in which organic matter contained in the mass of waste decomposes. The result of these processes is the release of biogas, which can be a real energy source. The calorific value of landfill biogas, depending on its chemical composition, ranges from 17-27 MJ/m^3. Landfill biogas includes among others CH_4, CO_2, N_2, and traces of H_2S, CO or NH_3, of which only methane and hydrocarbon gases (C_nH_m) are relevant for energy efficiency.

In 2018, there were 286 municipal landfills in Poland, 258 of which were equipped with a degassing installation, of which 23 were equipped with a degassing installation with heat recovery and 68 with a degassing installation with electricity recovery (the remaining ones were equipped with degassing installations without energy recovery). Nearly 85 thousand GJ of heat and over 105 thousand MWh of electricity were produced from biogas storage sites (https://www.teraz-srodowisko.pl/aktualnosci/gaz-skladowiskowy-skladowiska-odpadow-metan-biogaz-7800.html).

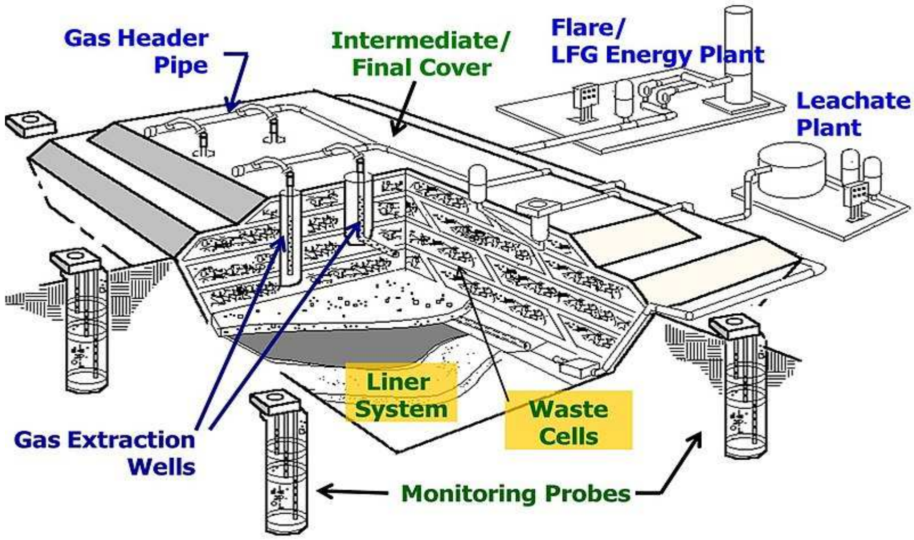


Fig. 4.10. Ideological diagram of biogas extraction from a landfill (https://commons.wikimedia.org/wiki/File:Landfill_gas_collection_system.JPG)

In many EU countries, anaerobic digestion installations have been authorised to inject biomethane into national gas networks. Biomethane is produced from biogas in a purification and upgrading process that removes pollutants and CO_2 from the biogas to meet gas network standards. Biomethane is fed into the natural gas network through a pipeline, the cost of which usually represents one of the highest installation costs (Baccioli, 2019).

The use of biofuels for transport started in Brazil in the early 20th century and increased in the 1970s after the first oil crisis. Later, biofuels appeared as an alternative to fossil fuels to reduce greenhouse gas emissions from the transport sector. In 2015, 132 billion litres of biofuels were used in transport worldwide. Of this, 98 billion are ethanol, 30 billion are biodiesel, and 5 billion litres of hydrogenated vegetable oils. The world leader in the use of biodiesel in transport is the European Union (12 billion litres) (Scarlat, 2018).

4.4. SUMMARY

Current regulations are increasingly strict about the problem of sewage sludge storage in overcrowded municipal landfills. Due to the increasing possibilities of expansion and modernization of existing domestic sewage treatment plants or the emergence of new facilities, in the coming years, Poland will see a sharp increase in sewage sludge stream, the management of which will be quite challenging. The still popular natural methods of sewage sludge management are also losing their attractiveness as they contain large amounts of heavy metal compounds that are harmful to the environment in municipal sewage sludge from large urban agglomerations in particular. Therefore, it seems that the target direction of sewage sludge utilization in the future will be primarily thermal disposal methods, as environmentally safe and economically justified. Sewage sludge is an attractive alternative fuel that can be used to produce electricity and heat.

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Chapter 5

Creation of a systematic, descriptive review of the main sewage sludge/biowaste processing technologies

Katarzyna WYSTALSKA

5.1. INTRODUCTION

Biodegradable waste from the agri-food industry and municipal economy, as a result of its physical-chemical nature, requires proper treatment, which not only eliminates its negative environmental impact but also allows for recovery of raw materials and energy. Such actions are one of the crucial elements of the circular economy. Economic development, growing consumerism, increased "production" of waste with a tightening legal discipline regarding the waste management (the need to selective collect the communal biowaste fraction) determine the use of specific methods of its processing.

Biodegradable waste (in accordance with the Act on Waste) includes waste susceptible to anaerobic and aerobic digestion using microorganisms; whereas biowaste, according to the Act, is "biodegradable waste from gardens and parks, food and kitchen waste from households, gastronomy, mass catering establishments, retail units, and comparable waste from plants producing or distributing food to the market", a separate group is formed by waste from agriculture and forestry, sewage sludge, excrements, natural fibers and paper. Although this waste group is excluded from the statutory definition of biowaste, it remains biodegradable waste.

The tightening up of regulations on the selective collection of waste, including biodegradable waste results in, among others, the increase in the communal biowaste fraction. Such actions also create the need to use a proper method of its management.

The main technologies used for treatment of this type of waste are anaerobic stabilization (methane fermentation), aerobic stabilization (composting), and thermal methods (Fig. 5.1). Anaerobic stabilization is mainly used in wastewater treatment plants to stabilize sewage sludge and agricultural biogas plants, where the by-products from agri-food production and animal husbandry are processed, but it is also used to stabilize the biodegradable fraction of communal waste. Aerobic

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stabilization, i.e., composting, is used mostly to process the biodegradable fraction of communal waste and is carried out more often in bioreactors or compost piles in single- or two-step systems.

BIODEGRADABLE WASTE

Methane fermentation (Anaerobic stabilization)	Composting (Aerobic stabilization)	Thermal methods
Wet	A one-step	Combustion Co-combustion
Dry	Two-stage	Pyrolysis Gasification

Fig. 5.1. Methods used to process biodegradable waste

The use of thermal methods is considered primarily for sewage sludge and can be carried out through combustion, co-combustion, pyrolysis, and gasification processes. Methods such as wet oxidation, supercritical water oxidation, and hydrothermal methods are also known (Neczaj, 2016). The first two methods, i.e. the combustion and co-combustion are used in existing industrial infrastructure facilities, while pyrolysis and gasification are alternative methods the use of which is not yet as widespread.

5.2. METHANE FERMENTATION

One form of biodegradable waste stabilization, including sewage sludge, is anaerobic stabilization by methane fermentation. Methane fermentation is a very complex biochemical process that takes place under anaerobic conditions and leads to the complete or partial transformation of organic compounds into gaseous products (mainly CH_4 and CO_2), in production of intermediate products, such as alcohols or fatty acids (Bień et al., 2015; Bień and Wystalska, 2011; Janosz--Rajczyk, 2000). The process also involves microbiological reduction of sulphates to sulphides and hydrogen sulphide and anaerobic ammonification and reduction of nitrates (Janosz-Rajczyk, 2000). In simple terms, it can be said that methane fermentation occurs in four phases (Fig. 5.2): hydrolysis, acidogenesis, acetogenesis and methanogenesis (Bień et al., 2015; Bień and Wystalska, 2011).

In the hydrolysis phase, insoluble organic polymers (proteins, lipids, carbohydrates) are decomposed into soluble monomers (amino acids, simple sugars, polyhydroxy alcohols) and fatty acids, with the participation of hydrolytic enzymes (Fig. 5.2) (Neczaj, 2016).

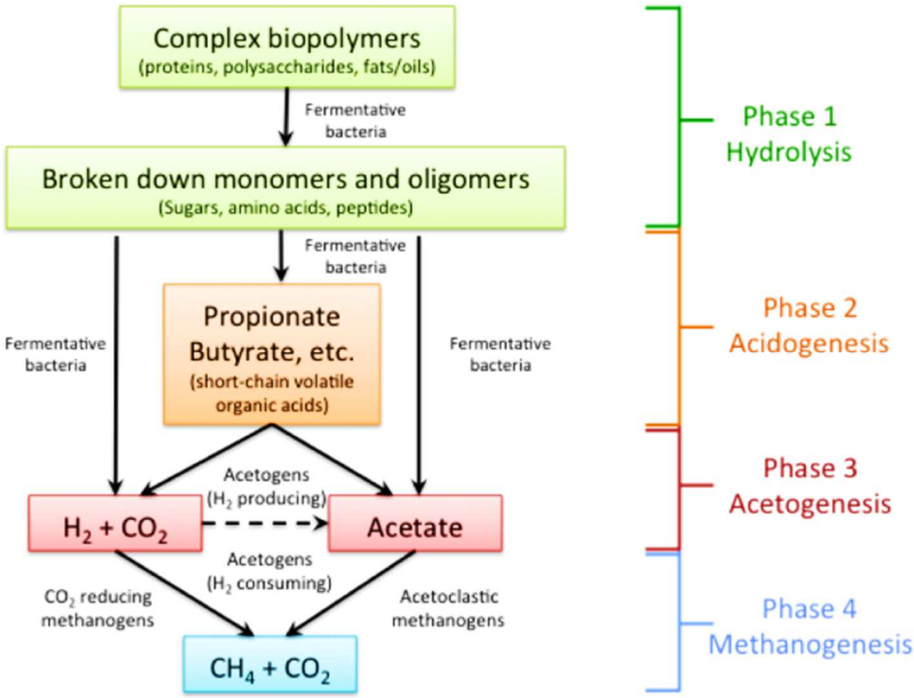

Fig. 5.2. Methane fermentation phase (https://www.e-education.psu.edu/egee439/node/727)

This phase is a difficult stage affecting the speed and effects of the entire process. (Myszograj, 2017). In the next phase, that is acidogenesis, hydrolysis products are broken down by facultative and obligate acidogenic bacteria into organic acids (formic, acetic, propionic, butyric, valeric, caproic), alcohols (methanol, ethanol), aldehydes and CO_2 and hydrogen (Neczaj, 2016). During acetogenesis, ethanol and volatile fatty acid (VFA) are processed into acetate and CO_2 and H_2 by acetogenic bacteria. These are the bacteria that live in symbiosis with other species between which there is the so-called hydrogen transfer. A prerequisite for the transitions in this phase is low pressure of hydrogen, which is also used by sulfate reducing bacteria (Neczaj, 2016). Methane production by methane bacteria takes place in the methanogenesis phase.

Methane is produced in the following ways (Neczaj, 2016):

- as a result of acetic acid decomposition led by heterotrophic bacteria according to the equation:

$$CH_3COOH \rightarrow CH_4 + CO_2$$

- as a result of CO_2 reduction and in a small percentage from dissolved hydrogen

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$$

- from the following substrates:

$$4HCOOH \rightarrow CH_4 + 3CO_2 + 2H_2O$$

$$4CH_3OH \rightarrow 3CH_4 + CO_2 + 2H_2O$$

$$4CO + 2H_2O \rightarrow CH_4 + 3H_2CO_3$$

$$4(CH_3)_3 N + 6H_2O \rightarrow 9CH_4 + 3CO_2 + 4NH_3$$

$$2(CH_3)_3 NH + 2H_2O \rightarrow 3CH_4 + CO_2 + 2NH_3$$

$$4(CH_3)_3 NH_2 + 2H_2O \rightarrow 3CH_4 + CO_2 + 4NH_3$$

$$2(CH_3)_2 S + 3H_2O \rightarrow 3CH_4 + CO_2 + N_2S$$

$$4Me^0 + 8H^+ + CO_2 \rightarrow 4Me^{++} + CH_4 + 2H_2O$$

5.2.1. METHANE FERMENTATION OF SEWAGE SLUDGE

In wastewater treatment plants, in the process of methane fermentation, thickened primary and excess sludge thickened are the primary sludge and excess sludge that are often subjected to the disintegration process to intensify the hydrolysis project through the change in the share of insoluble fraction and the structure of organic compounds (Myszograj, 2017).

Among the methods of disintegration, the distinguishable ones include thermal disintegration (40-180°C), chemical disintegration (ozone, acids, alkali), mechanical disintegration (ultrasound, mills, homogenizers), biological disintegration (action of enzymes), freezing and thawing of sediments, advanced oxidation processes (Myszograj, 2017).

Methane fermentation is brought to the so-called technical fermentation limit which occurs when 40-50% of organic substance has been reduced from sludge subjected to that process (Bień and Wystalska, 2011). Fermented sludge is then subjected to the dewatering process and the final management process (Fig. 5.3).

Fermentation depending on the used temperature is carried out as psychrophilic, mesophilic, or thermophilic fermentation (Bień and Wystalska, 2011). Frequently, multi-stage systems are used that can reduce stabilization time, ensure sediment hygiene, and pose less risk of process disruption (Wójtowicz et al., 2013).

The devices in which the methane fermentation process is carried out are closed fermentation chambers. These chambers are made in the form of steel or reinforced concrete tanks, with a capacity most often in the range of 1000-8000 m^3 (Bień and Wystalska, 2011). These tanks are made in the shape of a cylinder with truncated cones at the top and bottom and are egg-shaped (Fig. 5.4) (Bień et al., 2014). To ensure proper functioning of the fermentation chamber, it must be equipped, among others, in equipment for heating sludge, mixing chamber contents, compression and discharge systems for the produced biogas (Bień and Wystalska, 2011).

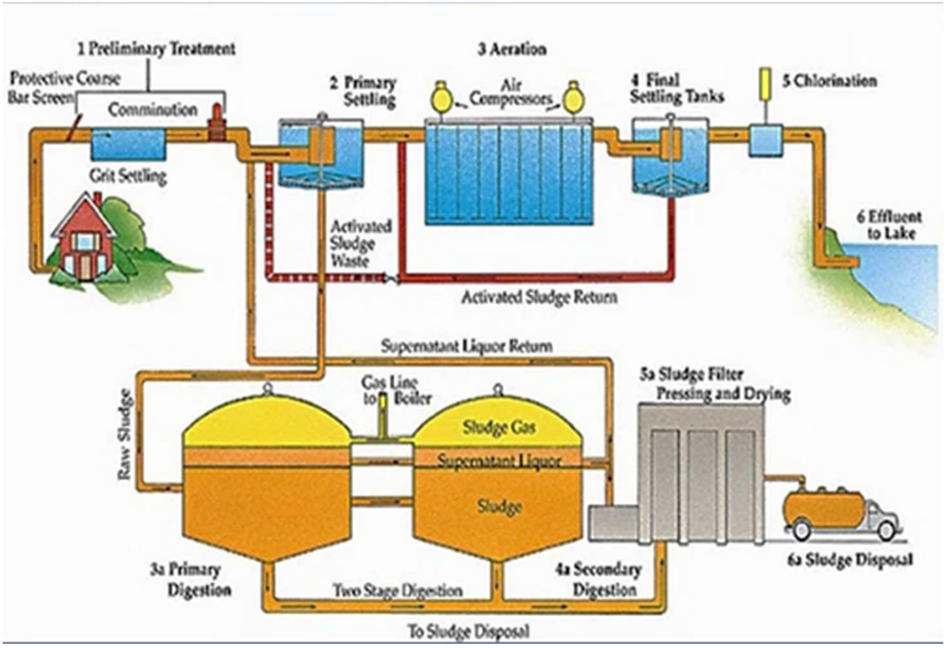

Fig. 5.3. Place of methane fermentation in the technological scheme of wastewater treatment plant (https://www.indiamart.com/proddetail/sewage-treatment-plant-stp-5357042088.html)

a) b)

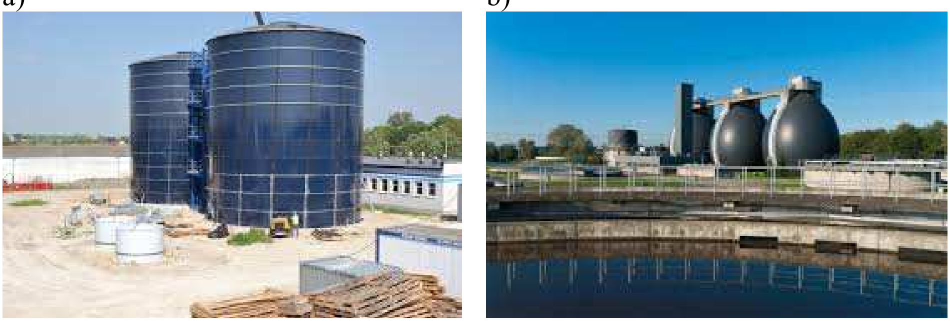

Fig. 5.4. Closed fermentation chambers: a) cylindrical, (http://www.jrp.mpgk.chelm.pl/aktualnosci/zamknite-komory-fermentacyjne), b) egg shaped (http://e-czytelnia.abrys.pl/wodociagi-kanalizacja/2013-12-725/oczyszczanie-sciekow-8442/oczyszczalnia-sciekow-stabilizacja-osadow-cz-v-17515)

5.2.2. METHANE FERMENTATION IN AGRICULTURAL BIOGAS PLANTS

The process of methane fermentation is also used in agricultural biogas plants, where biodegradable substrates, mainly from the agri-food industry, are processed into biogas used to manufacture heat and electrical energy (Fig. 5.5). After the process, what is left is the so-called digestate that can be used as a substitute of the fertilizer.

Raw materials used in biogas plants can be classified into two main groups: plant biomass from field crops, grassland, water bodies and others, and biomass from by-products and waste arising from the economic activities of different units (Podkówka, 2012). Biomass in the second group, according to the authors (Podkówka, 2012), should contain > 30% of the organic substance. Raw materials used in biogas plants include by-products and waste from food processing of plant and animal raw materials, green waste from agriculture, fruit culture and forestry, household waste, communal waste, waste from wood processing, furniture and chipboard production, sewage sludge, animal waste, paper, paperboard, and waste from pharmaceutical and cosmetic industries (Podkówka, 2012). In biogas plants, waste and by-products are used first, and only then the special-purpose crops (http://www.argoxee.com.pl/biogazownie.php).

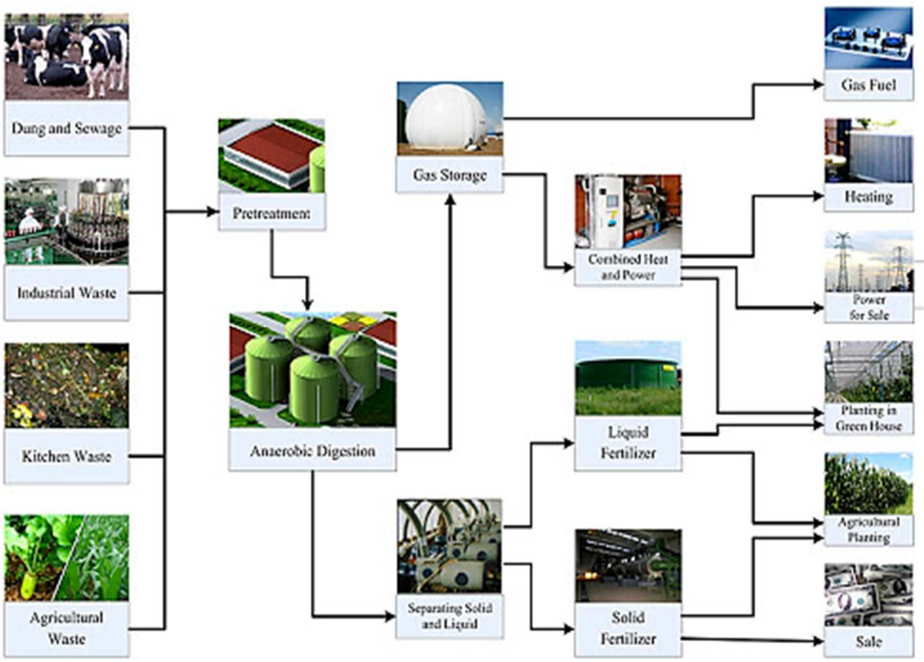

Fig. 5.5. Biogas production and use in agricultural biogas plants (http://biofuelsacademy.org/index.html%3Fp=1595.html)

In biogas production technology in agricultural biogas plants, the stage of feedstock preparation is essential. At this stage, the processes of sorting, hygienization and trituration (Podkówka, 2012) are distinguished. Sorting is intended to remove non-biodegradable and disruptive materials, e.g. stones and other impurities. In the case of biogas plants that use animal by-products requiring hygienization, in this stage, the mass is heated before entering the fermenter at 70°C for at least an hour. The chamber feedstock (e.g. vegetables, straw, manure) often requires trituration, which facilitates fermentation and increases its efficiency (Podkówka, 2012).

The prepared feedstock is fed into the fermentation chamber by means of pumps (for liquid substrates) or wheel loaders (for solid substrates) (Podkówka, 2012). In agricultural biogas plants, the fermentation process is most often carried out as wet fermentation, semi-dry and dry fermentation are much less common (Podkówka, 2012). To avoid stratification of the feedstock and ensure the proper course of the fermentation process, the contents of the fermentation chamber are mixed and heated by the suitable installation (Podkówka, 2012).

The properly conducted fermentation process requires an optimal load of the chamber with organic pollutants, an appropriate hydraulic retention time, adapted to the type of feedstock, ranging from about 20 days for liquid manure to about 60 days for energy plants, adequate temperature and elimination of process inhibitors such as antibiotics and plant protection products that delay the process (http://www.argoxee.com.pl/biogazownie.php).

5.2.3. METHANE FERMENTATION OF SOLID WASTE

Methane fermentation depending on feedstock hydration can be carried out as a wet process (6-15% of dry mass), semi-dry (20±2% of dry mass), or dry (> 20±2%, but < 45% of dry mass) (Podkówka, 2012). According to Jędrczak (2008), the dry matter content above 40% can lead to the inhibition of biological processes. Single-stage dry fermentation is an economic method that reduces the volume of the process reactor (compared to wet fermentation) and can be used to transform a properly fragmented (2-4 cm) under-sieve fraction of communal waste (Sidełko 2018).

Dry fermentation is carried out in plug flow reactors which differ in the way of mixing the feedstock in the reactor (Jędrczak 2008). The features of the reactors used for this process are the way in which organic mass is moved inside and outside the reactor, as part of the recirculation, and the heat necessary to achieve optimal temperature conditions associated with the intensity of biochemical transformations (Sidełko and Chmielińska-Bernacka, 2013). Technical reactor solutions are based on technologies developed by Dranco (Fig. 5.6), Kompogas and Valorga (Jędrczak, 2008). In the Dranco vertical reactor, the feedstock is delivered to the top of the pump and then gravitationally moves to the bottom of the reactor. In the Kompogas reactor, which has a cylindrical shape and is horizontally positioned, there are axially positioned agitators causing homogenization and degassing of the feedstock. Valorga reactors are using a stream of biogas injected under pressure to the moving waste (Sidełko, 2018; Sidełko and Chmielińska-Bernacka, 2013).

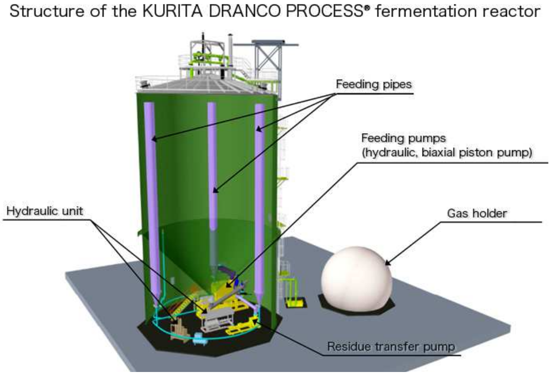

Fig. 5.6. Reactor used in dry fermentation (https://www.kurita.co.jp/english/aboutus/press170202.html)

According to Sidełko (2018), in communal waste treatment plants with a capacity of more than 15 thousand tons per year, methane fermentation is a more cost-effective method of stabilization compared to composting. In operating installations, the amount of energy generated from biogas produced exceeds the energy demand of equipment related to this technology (Sidełko, 2018).

5.2.4. CO-DIGESTION

Co-digestion is a simultaneous transformation into biogas of a homogeneous mixture of two or more types of biomass (Klimiuk et al., 2012). The tendency to implement this type of fermentation process was provoked by problems related to the sensitivity of methanogens to high ammonia concentrations accompanying the conversion of animal excrement or too low a level of nutrient concentrations often occurring in the transformation of plant biomass (Klimiuk et al., 2012). Co-digestion ensures the right proportions of organic compounds and nutrients or structural components, improving the rheological properties of the feedstock and a decrease in the content of process inhibitors (Klimiuk et al., 2012). The appropriate selection of the C/N ratio prevents the fermentation process from stopping (Waited et al., 2017). Substrates that are most commonly used in co-digestion with sewage sludge are fatty and agri-food waste and distillery grains (Czekała et al., 2017). According to the authors (Klimiuk, 2012), the addition of inadequate co-substrate can cause deterioration of the process or enforce the need to equip the facility with additional infrastructure. However, a properly implemented co-digestion process may not only contribute to the absence of technological distortions in the process, but may also significantly increase biogas production (Podkówka, 2012), or increase the methane content of biogas (Bień et al., 2010; Gazda et al., 2012; Grosser et al., 2013).

5.2.5. FACTORS INFLUENCING THE METHANE FERMENTATION PROCESS AND CONTROL PARAMETERS

Factors that decisively affect the course and effects of fermentation include organic matter load in the chamber, process temperature, mixing of chamber contents, pH, toxic content (Bień and Wystalska, 2011; Bień et al., 2015; Neczaj, 2016).

The load of the fermentation chamber is expressed by the amount of substrate weight introduced to 1 m^3 of volume per day and is most often expressed in kg of dry organic mass/$m^3 \cdot d$ (Podkówka, 2012). The optimal load enabling the fermentation process to run properly in agricultural biogas plants is approximately 5 kg of dry organic mass/$m^3 \cdot d$ (Podkówka, 2012). In wastewater treatment plants, during methane fermentation of sewage sludge, the frequency of supply of fermentation chambers with sediments affects the load on the chamber by organic pollutants. Therefore, it depends on the fermentation temperature and the size of the treatment plant (Bień and Wystalska, 2011).

According to Neczaj (2016), the process temperature affects the rate of enzymatic reactions, mass transport, and solubility of chemical compounds. Large temperature fluctuations (greater than 2°C) can lead to disorders in the process, and its rapid changes (10°C) to the death of bacteria (Bień and Wystalska, 2011). Methane fermentation can be conducted under psychrophilic conditions (< 20°C), mesophilic (30-40°C), thermophilic (45-60°C) and extremely thermophilic (> 60°C) (Bień and Wystalska, 2011; Neczaj, 2016). Most often, however, the process is carried out under mesophilic conditions.

Mixing the contents of the fermentation chamber is very important for the proper course of this process; it intensifies the decomposition processes, prevents the decomposition of the feedstock (formation of sludge blanket), facilitates the discharge of the biogas produced, ensures an even concentration of metabolism products, and prevents the formation of zones underheated in the chamber (Bień and Wystalska 2011).

The reaction of the fermentation chamber content is a function of the presence of acids: carbon, phosphoric, volatile organic acids, hydrogen sulfide and ammonium hydroxide (Podkówka, 2012). The bacteria responsible for the fermentation process require a pH in the range of 6.8-7.4; for pH < 6 and pH > 8, there are process disturbances (Podkówka, 2012).

Toxic substances that may disrupt the process include (Neczaj, 2016) gas ammonia, hydrogen sulfide, sulfides, toxic organic compounds. The action of fermentation inhibitors depends on the form in which they occur and their concentration (Podkówka, 2012). Light metal cations, at appropriate concentrations, have a stimulating effect on the fermentation process, but in higher concentrations, they can have an inhibitory effect. Their form is also important because only ionic metal forms are toxic (Podkówka, 2012).

For carrying out the methane fermentation process in biogas plants, essential factors influencing the process are also butyric acid bacteria (*clostridium*) present in low-quality ensilage (Podkówka, 2012). This group of bacteria forms spores classified as anaerobes. In the process of fermentation of the ensilage, these bacteria form butyric acid from carbohydrates and amines and ammonia from proteins. These compounds inhibit the process of methane fermentation. Another factor is the mold fungus producing toxins with antibacterial activity also against the bacteria of methane fermentation (Podkówka, 2012).

For stability and proper course of the process have an influence three strongly related parameters: pH, alkalinity, volatile fatty acids (VFA). As reported by the (Dymaczewski, 2011), the pH limits in the fermentation process are 6.5-8.5 and the limit of the VFA content (expressed as acetic acid) should not exceed 2000 mg3COOH/L. For a properly carried out fermentation process, the pH of the supernatant is neutral (pH = 7-7.2), with a simultaneous VFA content of 100--500 mg/L and alkalinity not less than 500 mg $CaCO_3$/L (Dymaczewski, 2011).

5.2.6. BIOGAS PRODUCED BY FERMENTATION

Biogas produced by the methane fermentation process is a valuable product that can be used to produce heat and electricity. The main component of biogas is methane, although in smaller quantities there is carbon dioxide, nitrogen and oxygen (Table 5.1). The methane content of biogas determines its caloric content. It is assumed that biogas containing approximately 65% of methane has a caloric value of 23 MJ/m^3 (http://www.mae.com.pl/files/poradnik_biogazowy_mae.pdf). The quantity and composition of the biogas depends on the type and share of the individual substrates converted. The addition of fats to the fermentation chamber feedstock may increase the biogas production, while proteins can produce biogas with a high methane content. In biogas obtained from carbohydrates, the amount of methane is approximately 50% (Podkówka, 2012).

Table 5.1

Biogas composition (http://www.mae.com.pl/files/poradnik_biogazowy_mae.pdf)

Ingredient	Volume share, %
CH_4	50-75
CO_2	25-45
N_2	< 2
O_2	< 2
H_2	< 1
CO	0-2.1
H_2S	20-20 000 ppm
others	trace amounts

The presence in biogas of components such as hydrogen sulfide, water vapor or sludge particles reduces its energy value and may negatively affect the functioning of equipment (corrosion of armature, piping, tanks) (http://www.mae.com.pl/files/poradnik_biogazowy_mae.pdf). To remove the impurities, biogas is subjected to purification processes, consisting mainly in desulfurization, removal of fine impurities and drying (Bień and Wystalska, 2011; http://ksow.pl/fileadmin/user_upload/ksow.pl/pliki/zaproszenia/R%C3%B3%C5%BCne/publikacja_Biogazownie.pdf).

The production of the biogas, due to the availability of raw materials (agricultural biogas plants) or their quality, may fluctuate temporarily. To compensate for the unevenness of production, biogas is stored in appropriate tanks. Tanks used for this, due to the prevailing pressure, are divided into low-, medium-, and high-pressure tanks (Bień and Wystalska, 2011). With regards to the type of the seal, wet and dry tanks are distinguished (Bień and Wystalska, 2011). In agricultural biogas plants, often used are external low-pressure vessels in the form of foil pillows or mounted directly on the bioreactor in the form of the so-called foil domes (Fig. 5.7), hermetically attached to the top edge of the tank (Podkówka, 2012).

Fig. 5.7. Fermentation tanks in an agricultural biogas plant – view of the fermentation tanks (https://poranny.pl/biogazownia-w-rybolach-pracuje-juz-pelna-para/ar/10183410)

Wastewater treatment plants most often use external tanks for biogas storage (Fig. 5.8).

Biogas caloric content allows it to be used for the following purposes (Podkówka, 2012):

- production of electricity in spark-ignition engines and turbines,
- production of heat in gas boilers,
- production of electricity and heat in cogeneration units,
- as fuel for vehicle engines,
- for methanol production,
- use in the natural gas network.

Fig. 5.8. Biogas tank (http://www.sigatech.pl/pl/oferta/wyposazenie-sieci-biogazu/zbiorniki-magazynowe-biogazu.html)

The production of biogas in wastewater treatment plants or agricultural biogas plants enables supplying these facilities in heat necessary for heating buildings or technological facilities and allows for the production of electricity, which can be sold to proper companies. Biogas is most often processed in cogeneration engines (Fig. 5.9) that produce energy and heat. The excess of biogas produced, for example, during the summer period, can be burned in the case of an emergency in the so-called biogas flares (Bień and Wystalska, 2012).

Fig. 5.9. Cogeneration unit (https://www.instalator.pl/2018/04/agregaty-kogeneracyjne-w-biogazowni/)

Engines with compression ignition or spark ignition are most often used as Combined Heat and Power (CHP) units allowing for combined energy economy. An alternative for them may be the use of gas turbines, stirling engines or fuel cells; however, currently they are of lesser importance (https://www.instalator.pl/2018/04/agregaty-kogeneracyjne-w-biogazowni/).

5.3. COMPOSTING

Composting involves an aerobic decomposition of organic matter with the participation of microorganisms (Jędrczak, 2008). The process yields a solid product – compost and carbon dioxide, water and heat (Jędrczak, 2008). The purpose of the composting process is to obtain a stable material classified as organic fertilizer or a stabilized product (Sidełko, 2018). In composting, very complex and prolonged biochemical transformations occur (Fig. 5.10), during which organic matter is transformed due to mineralization and a second synthesis of organic polymers (mainly humus) occurs where a second synthesis of organic polymers mainly humus compounds occurs (Sidełko, 2018).

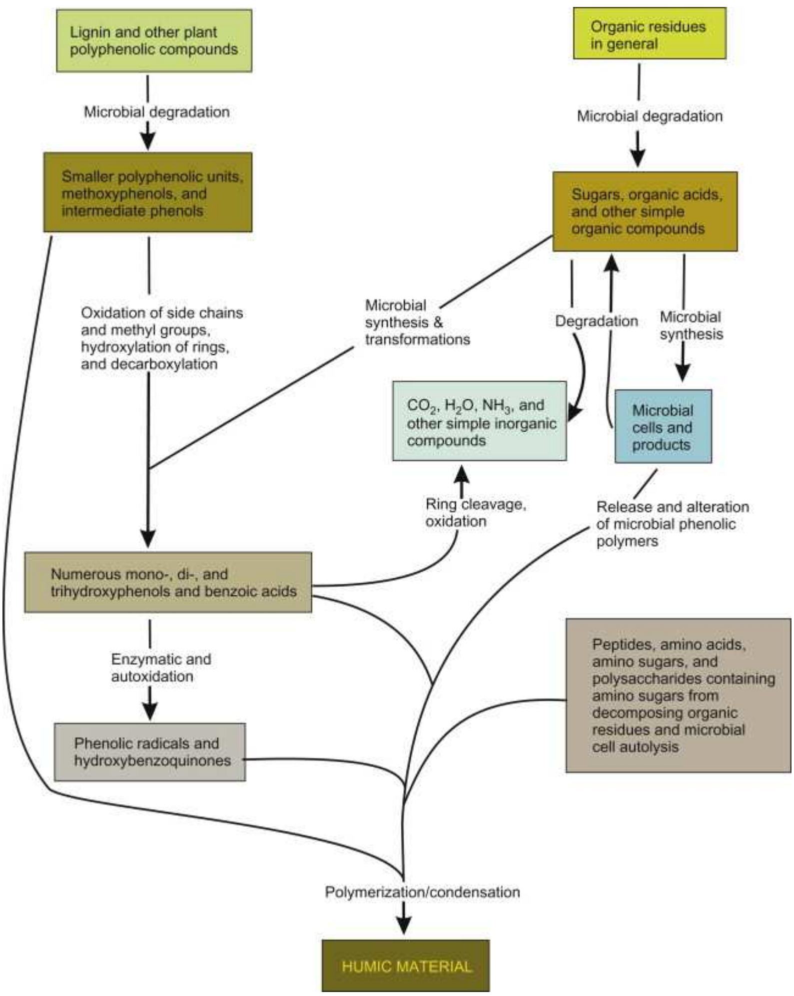

Fig. 5.10. Scheme of decomposition of organic matter and synthesis of humus compounds (https://www.sciencedirect.com/topics/earth-and-planetary-sciences/humus)

Under optimal conditions, composting takes place in four phases, differing in the activity of microorganisms (Jędrczak, 2008):

1. Pre-composting phase, mesophilic phase, lasting several days during which temperature rises (Fig. 5.11),
2. Intensive composting phase (high-temperature phase), thermophilic phase, lasting from a few days to several weeks, during which the easily biodegradable compounds are decomposed, water, carbon dioxide, and ammonia are formed,
3. The phase of transformation (proper composting), lasting about 3-5 weeks, during which there is a decrease in temperature and a transformation of compounds such as lignin, fats, wax, resins by mesophyll bacteria and fungi,
4. The compost maturation phase (secondary composting) lasts several months, during which temperature lowering and the formation of a stable compost fraction (humus) occurs, macrofauna also appears.

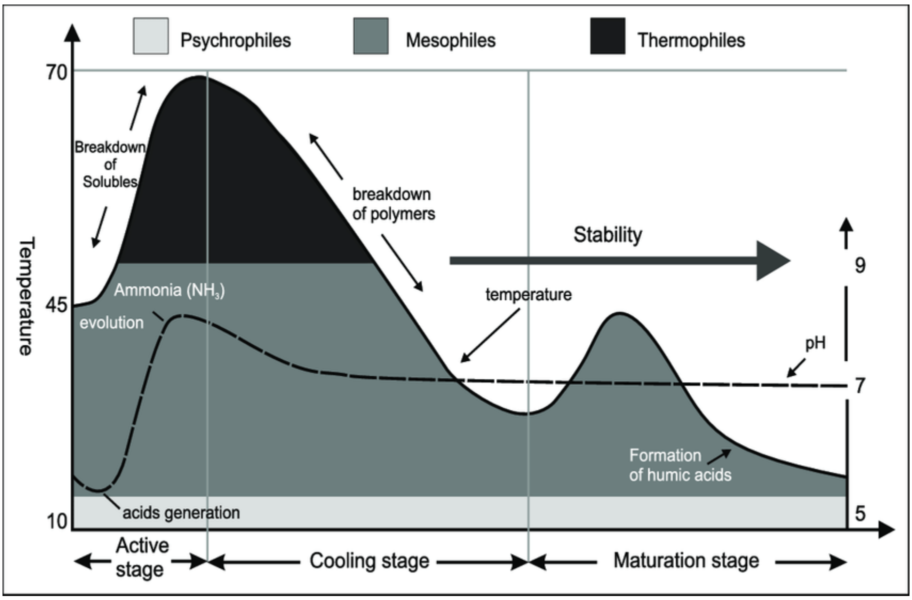

Fig. 5.11. Temperature changes in the different composting phases (https://www.researchgate.net/figure/Composting-phases-modified-from-Biddlestone-and-Gray-1985_fig2_47932434)

During the composting process, parameters such as feedstock structure, water content, nutrients content, aeration, process duration, temperature, pH are important (Sidełko, 2018; Bień et al., 2015; Jędrczak, 2008).

The structure of the composted mass is associated with the dimensions of the individual particles forming the feedstock, which directly affects the amount of water and air contained in the pores of the material. Less porosity promotes water filling of space between molecules, and increased humidity inhibits the activity of microorganisms, which can generate anaerobic processes of decomposition of organic matter (Sidełko, 2018). The share of air-filled space in the pores of composted material varies between 10 and 40%, and this range corresponds to a change in humidity in the range of 55-90% (Sidełko, 2018). By using appropriate treatments, i.e. addition of structural material or trituration, the porosity of the composted feedstock may be changed. Particles too large in size limit the surface of contact with the microorganisms responsible for the distribution of the matter. Particles too small in size can increase airflow resistance; therefore, the optimal size of the composted feedstock should be in the range of 2-6 cm (Sidełko, 2018). Jędrczak (2008) defines the porosity of the composted material as the quotient of pore volume and the total volume of the material. The free air space (FAS) index binding bulk density and the specific density of the composted material (Bień et al., 2015) is used to assess porosity. For waste with an optimal humidity in the range of 45-65%, FAS values range from 36 to 50% (Jędrczak, 2008). It is assumed that the FAS index should correspond to a minimum of 30% of the volume of composted material, and the dimension of the substrate intended for composting, at optimum humidity, should ensure the share of the volume of free air space in the range of 25-35% (Jędrczak, 2008).

The water content significantly affects the correct course of composting as the humidity is a factor limiting the intensity of biochemical transformations, resulting in the temperature being maintained at an appropriate level (Sidełko, 2018). The amount of water that is released due to the activity of microorganisms is greater than the loss as a result of evaporation; however, some of the water is removed from the compost reactors as a result of aeration (Sidełko, 2018). According to Sidełko (2018), the optimal moisture content of the compost mass should be in the range of 50-60%. According to Jędrczak (2008), composting takes place at a good speed when the moisture content is in the range of 45-60%. However, with humidity above 65%, the oxygen flow in the waste layer is limited and anaerobic areas are formed (Jędrczak, 2008). Water is the factor which determines the intensity of the processes of aerobic and anaerobic decomposition of organic matter, which run in parallel during composting (Sidełko, 2018). When the moisture content falls below 40-45%, microorganisms have difficulty in accessing the nutrients, their activity decreases, and the composting process begins deaccelerating significantly, while with moisture at less than 20%, composting practically does not occur (Jędrczak, 2008).

The presence of nutrients (C, N, P) in the compost mass affects the development and growth of microorganisms. Carbon is the primary source of energy, nitrogen determines the growth of microbes, and phosphorus and potassium are important for cell metabolism and reproduction (Jędrczak, 2008). The proportions between the different nutrients are essential. The C/N and C/P ratios are the basic criteria for assessing the susceptibility of a material to aerobic biological decomposition (Sidełko, 2018; Jędrczak, 2008). Too low C/N content (< 25:1), means a high concentration of nitrogen and its excess, as a mineral form N-NH_3, can inhibit the growth of microorganisms and cause odor nuisance (Sidełko, 2018; Jędrczak, 2008). A C/N ratio that is too high (> 50:1) means a low nitrogen content and limited development of microorganisms (Sidełko, 2018; Jędrczak, 2008). For the correct course of the composting process, the C/N ratio should be in the range from

25:1 to 35:1 (Jędrczak, 2008). The value of the C/N ratio of the composted biomass can be controlled by the addition of appropriate components with a low or high C/N content (Jędrczak, 2008). Examples of such materials are sawdust (C/N = 500), old paper (C/N = 300), wheat straw (C/N = 128), sewage sludge (C/N = 15) (Jędrczak, 2008). During composting, the C/N ratio is reduced as organic carbon is converted to CO_2. For mature compost, this value is 10-15/dm^3 (Jędrczak, 2008). The addition of structuring material not only affects the value of the C/N ratio, but also the moisture level of the composted biomass. Composting materials characterized by high humidity (sewage sludge or animal droppings) and a low C/N ratio simultaneously, require the use of structuring agents (straw, wood chips, sawdust, rice husks, rice bran, wood chips, hay, and others) (Bień et al., 2015). Materials in the form of polypropylene discs can be used as fillers, which are removed while sieving the mature compost (Bień et al., 2015).

Aeration of the composted biomass is carried out to ensure the biological activity of microorganisms and to remove excess moisture and heat (Jędrczak, 2008). The right amount of air inhibits anaerobic processes and regulates the heat balance during composting (Sidełko, 2018). A measurable effect of aeration is the rate of growth and the achieved temperature value; therefore, the amount of supplied air should be adjusted to the dynamic conditions in the bioreactor (Sidełko, 2018). The amount of oxygen supplied to bioreactors is usually in the range of 0.6-1.9 m^3/kg cl within 24 hours (Sidełko, 2018). According to Jędrczak (2008), the oxygen concentration should be between 12 and 21% (v/v) (optimal range > 15%). At the beginning of the composting process, aeration should be intensive, and it should be reduced during the maturing phase (Jędrczak, 2008).

Depending on the method of composting used, the duration of the process varies. According to Sidełko (2018), one-stage composting in the compost piles, carried out to obtain the mature compost, requires about 20-25 weeks, while the use of a two-stage process takes about 7-21 days of the process in the bioreactor and 9-12 weeks of keeping the compost on the compost piles.

Temperature is a key parameter in the composting process for the proper decomposition of organic matter (Jędrczak, 2008). Thermophilic bacteria generate up to 4 Wh of energy per gram of oxygen consumed ($14.4 \cdot 10^6$ J/kg O_2) and during the intensive composting phase, about 1 MJ of heat is released from 1 kg of waste (Jędrczak, 2008). Composting is the fastest in the 45-55°C temperature range. It is very important that the temperature does not fall below 20°C because then the reproduction of microorganisms is inhibited and at the same time, it should not be higher than 60°C because that can limit the reproduction of actinomycetes (Bień et al. 2015; Jędrczak 2008). Changes in temperature and reaction during composting are shown in Figure 5.12.

The optimal pH value for bacteria of the composting process ranges from 6.0 to 7.5, while for fungi from 5.5 to 8.0 (Bień et al., 2015; Jędrczak, 2008). A decrease in pH below 6 leads to the death of microorganisms, while its increase above 9 causes nitrogen to convert into ammonia, which is released into the atmosphere, making it inaccessible to microorganisms (Bień et al., 2015; Jędrczak, 2008).

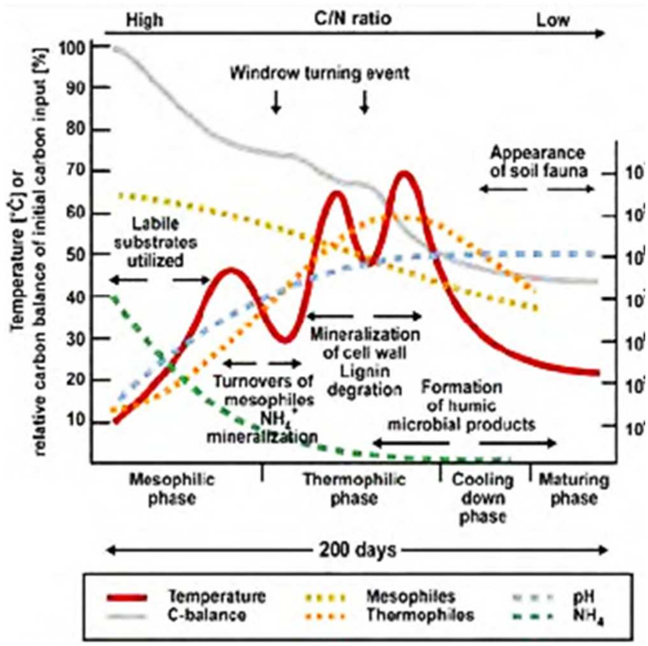

Fig. 5.12. The course of changes in temperature and reaction of the composted feedstock (https://www.researchgate.net/figure/Different-stages-during-composting-as-function-of-time-appearance-and-succession-of_fig1_221923455)

The product obtained in the composting process must meet quality requirements (specified in the applicable legal regulations) if it is to be used for fertilizing plants or soils or for reclamation. Quality standards concern, among others, the highest allowed content of heavy metals (Cr, Zn, Cd, Cu, Ni, Pb, Hg), the presence of pathogenic microorganisms and the minimum content of organic matter, nitrogen, phosphorus, and potassium (Bień et al., 2016).

5.3.1. AEROBIC STABILIZATION OF THE BIODEGRADABLE FRACTION OF COMMUNAL WASTE

Mixed communal waste that arrives at plants responsible for its treatment is subjected to mechanical and biological processing. Mechanical treatment mainly consists of segregation and sieving, which allow for the separation of a stream of raw materials suitable for material recycling and the so-called bio fraction that undergoes aerobic stabilization. The remaining waste stream is directed to storage. Waste subjected to biological stabilization should contain an adequate amount of organic matter so that it can be transformed in this process. The so-called under-sieve fraction (< 8 cm) of mixed communal waste, depending on the category of the area from which it is collected and the seasons contains varying amounts of organic matter; this determines the quality of the product obtained during its stabilization. The physicochemical composition of this fraction makes it practically impossible to produce high quality compost (Sidełko, 2018) and, therefore, the product of aerobic stabilization is referred to as a stabilized compost and is most often used for reclamation of landfill slopes. The conditions for implementing this direction of stabilization are measured using the AT4 indicator, determining the amount of oxygen consumed by the biomass in four consecutive days, this value should be less than 10 mg O_2/g of dry mass (Sidełko, 2018).

Currently used technologies of aerobic stabilization of the biodegradable fraction of communal waste are based on two systems run in a single or two stages (Sidełko, 2018). The single-stage method consists of composting in the composting piles (Fig. 5.13), while in the two-stage method, the first phase of intensive composting is carried out in reactors and the compost matures in field conditions (piles) (Sidełko, 2018). The two-stage composting systems usually differ in the type of bioreactor used (Fig. 5.14) (Sidełko, 2018).

Fig. 5.13. Compost pile composting plant (https://www.aknova.com.pl/pl/Podstrony/Realizacje/Wykonawstwo/Kompostownia-w-RZGO-S-ajsino)

<table>
<tr><th colspan="12">COMPOSTING SYSTEMS</th></tr>
<tr><th colspan="7">Composting with the reactor</th><th colspan="5">Composting without a reactor</th></tr>
<tr><td>dynamic</td><td colspan="3">portioned with flow</td><td colspan="3">static</td><td colspan="3">static</td><td colspan="2">dynamic quazi</td></tr>
<tr><td>rotary</td><td colspan="2">horizontal</td><td>vertical</td><td rowspan="2">boxes</td><td rowspan="2">containers</td><td rowspan="2">tunnels</td><td colspan="2">hermetic</td><td>not hermetic</td><td>hermetic</td><td>not hermetic</td></tr>
<tr><td>drums</td><td>drums</td><td>tunnels</td><td>silos</td><td>stacks</td><td>heaps</td><td>flitch</td><td>flitch[1]</td><td>flitch[2]</td></tr>
</table>

[1] flitch – covered prisms

[2] flitch – prisms flipped uncovered

Fig. 5.14. Composting systems (Sidełko, 2018)

The unquestionable advantage of the two-stage systems is the reduction of uncontrolled emission of process gases including volatile organic compounds, ammonia, and hydrogen sulfide, accompanying the initial composting stage (Sidełko, 2018). The use of closed reactors in the first stage allows for optimization of process parameter, including aeration, which intensifies the composting process, it also allows purifying the discharged gases produced in the process (Fig. 5.15).

Fig. 5.15. View of the composting chambers (https://www.aknova.com.pl/pl/Podstrony/Realizacje/Wykonawstwo/Kompostownia-w-zak-adzie-CHEMEKO-SYSTEM-m-Rudna-Wi)

Composting in an aerated static composting pile is carried out for at least 21 days using pressure and vacuum aeration or a combination of both. The compost maturation stage lasts for about 30 days. The compost pile can be up to 5 m in height in such systems. Perforated pipes or ducts through which air is supplied are laid at the base of each pile. Compost piles are formed on the pipes covered with a layer of structuring material, the external surfaces of which are covered with the mature compost (Bień et al., 2015).

Composting in spread piles is the oldest composting system in which the stabilized material is regularly agitated for aeration. The piles have a height of 1.2 to 1.5 m, a length up to several dozen meters, and their width can extend up to 4 m. The intensive composting process is carried out from 4 to 6 weeks, during which the piles are agitated twice a week. Compost maturation takes place over the next 4-6 weeks (Bień et al., 2015).

During composting in aerated piles with agitation, forced aeration is supplemented with aeration due to agitation using special devices. This technology is characterized by a lower demand for land, a lower risk of odor emissions and a lower sensitivity to weather conditions (rain), but it is more expensive than the previously described method due to more complicated technical solutions (Bień et al., 2015).

5.3.2. COMPOSTING OF COMMUNAL SEWAGE SLUDGE

Sewage sludge contains many valuable substances easily absorbed by plants, but their direct use for fertilizing or improving soil properties is limited, among others, due to susceptibility to putrefaction, the presence of pathogens or its odorous properties. One of the possibilities of using sewage sludge is their stabilization in the composting process and then the use of compost as a soil improver.

The treatment of sewage sludge in the composting process can be treated (Podedworna and Umiejewska, 2008) as the only and basic process of stabilizing and sanitizing sewage sludge itself, as the second stage of sludge stabilization and the process of its hygienization, as a process of refining stabilized sludge and preparing material with high qualitative characteristics and a good structure.

Communal sewage sludge is difficult to compost because of its specific structure, high humidity, and a low C/N ratio. For this reason, it cannot be composted on its own, it requires the addition of structure-forming material that can optimize these parameters (e.g. straw, sawdust, chips) or the addition of other biowaste (Bień et al., 2013; Milczarek, 2015; Milczarek et al., 2013).

The legal regulations in force in Poland (Act on Waste, Minister of the Environment Regulation on the R10 recovery method) limit the possibilities of co-composting of communal sewage sludge with other biodegradable waste, e.g. the organic fraction of communal waste. The product obtained, regardless of quality, can only be used for land reclamation. Obtaining a product with good parameters (meeting the requirements of the Act on Fertilizers and Fertilization), which would constitute a commercial product, requires a lot of financial expenditures and the obtaining of certain certificates (Bień et al., 2015).

Examples of plants where sewage sludge is transformed into compost for sale are the wastewater treatment plants in Piła (http://gwda.pl/pl/kompostownia) and in Słupsk (https://www.wodociagi.slupsk.pl/dla-klientow-2/uslugi-dodatkowe/sprzedaz-kompostu/).

The Piła wastewater treatment plant composted and transported its/their own sewage sludge (http://gwda.pl/pl/kompostownia). Biodegradable waste from the food, paper and wood industry, agri-industry waste, green waste, and other waste the composition of which allows their use in the composting process is processed with the sludge. If necessary, the waste is comminuted with special equipment (http://gwda.pl/pl/kompostownia). To supplement the organic carbon content and humidity, straw, sawdust, shavings, bark, small chips, etc. are added to the compost mass. The composting process is carried out using the technology of agitated piles (Fig. 5.16), protected temporarily or permanently with specialized fabrics, forming a closed reactor. The process is monitored by measuring the temperature in the compost piles (http://gwda.pl/pl/kompostownia). The compost obtained contains no plastics, metals, hard materials, including pieces of glass, should not emit odors and has a temperature close to its surroundings. Mature compost must meet the parameters set out in the permits (decisions) granted to the manufacturer and is then sent out for sale. The recipients are farms and plants caring for green terrains and individual recipients using composts for their own needs. As the manufacturer emphasizes, compost as an organic fertilizer has a beneficial effect of loosening the structure of heavy soils, binding the light and sandy soils, as well as increasing the water and thermal capacity of soils (http://gwda.pl/pl/kompostownia).

The composting plant in Słupsk produces compost under the "Biotop" trade name (https://www.wodociagi.slupsk.pl/dla-klientow-2/uslugi-dodatkowe/sprzedaz-kompostu/). It is an approved organic fertilizer containing nutrients for plants. It can be used to improve the physical, biological, and chemical composition of soil in the basic plant production. It can also be used to shape urban green areas, lawns, sports fields, and golf courses, in forest management, cultivation of ornamental plants, nursery, communal management, covering and reclamation of degraded areas and landfills (https://www.wodociagi.slupsk.pl/dla-klientow-2/uslugi-dodatkowe/sprzedaz-kompostu/).

Fig. 5.16. View of the compost plant in the wastewater treatment plant Gwda (http://gwda.pl/pl/kompostownia)

The production of compost (carried out using the compost pile method) uses stabilized and dehydrated sewage sludge (approx. 32%), green waste from gardens, parks, and cemeteries in the form of tree branches, shrubs, leaves and green biomass (approx. 25%), straw (approx. 32%), waste bark as a structure-forming material (approx. 11%) (Sautycz, 2011).

5.4. THERMAL METHODS

Among the thermal methods used in the treatment of the sewage sludge are mono-combustion, co-combustion, pyrolysis, and gasification (Bień et al., 2014; Bień et al., 2015; Grobelak et al. 2019; Murakami et al., 2009; Pająk, 2013; Sobik-Szołtysek and Wystalska, 2019; Werle and Wilk 2010; Werle, 2014).

Mono-combustion is carried out in sewage sludge incineration plants (Bień et al., 2015; 2014; Pająk, 2013). As Pająk (2013) reports, this process is usually carried out using fluid technology (and in a few cases – grate technology) and provides the possibility of recovering thermal energy from combustion and using it to dry the sludge. It also allows for recovering phosphorus from ashes remaining after burning the sludge.

Co-combustion of sewage sludge can be carried out in communal waste incineration plants, power plants, combined heat and power plants and cement plants (Werle, 2014). In Polish communal waste incineration plants, however, no co-combustion of sewage sludge has been foreseen (Bień et al., 2014). Similarly, the energy industry is not prepared for such a method in terms of the infrastructure. Most solutions of this type operate in Germany (Pająk, 2013). Co-combustion of sewage sludge can be implemented in the cement industry (Sobik-Szołtysek and Wystalska, 2019), provided that the sludge meets certain quality standards as not to disturb the production technology.

Pyrolysis and gasification processes are not used on such a scale, as they remain in the sphere of technological research (Werle, 2014) or in using the obtained products, e.g. biochar (Ferreira et al., 2019; Czekała et al., 2019; Bavariani et al., 2019). Werle (2014), in his research, showed that gas from the sludge gasification can be used as a potential fuel for power boilers. However, this technology, as the author indicates, must be tested in terms of technical conditions and toxicity of the by-products.

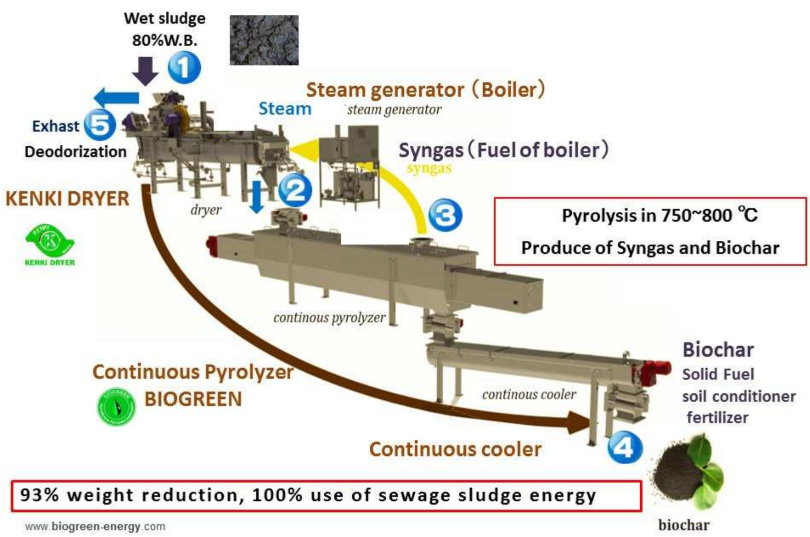

Fig. 5.17. Scheme of pyrolytic transformation of sewage sludge into biochar (https://kenkidryer.com/products/applications)

Pyrolysis appears to be a process that is more widely applicable to sewage sludge and other biodegradable waste (Fig. 5.17). It is a process of converting organic matter in anaerobic conditions and at high temperatures up to 1000°C (Wielgosiński, 2016). The products of this process are liquid (a mixture of oils, tar and water with simple aldehydes dissolved in it, alcohols, and organic acids), gas (pyrolysis gas containing mainly hydrogen, methane, ethane and their homologues, carbon monoxide and dioxide, and such compounds as: hydrogen sulfide, ammonia, hydrogen chloride and hydrogen fluoride) and solid (pyrolysis coke containing mainly carbon and other substances) products (Wielgosiński, 2016).

At present, pyrolysis is used to transform biodegradable waste into biochar (Dunnigan et al., 2018; Efika et al., 2018; Zhao et al., 2018; Zhao et al., 2018a). Plant waste can be used for biochar production, e.g. straw, wood chips, nut shells, sawdust, sewage sludge and chicken manure (Ahmed and Hameed, 2018; Bai et al., 2018; Batool et al., 2017; Bavariani et al., 2019; Kleemann et al., 2017). Biochar is at the center of attention of many researchers, primarily due to its specific properties and a wide range of potential applications. It can be used in engineering or environmental protection, among others, as an additive increasing the efficiency of the composting process (Agyarko-Mintah et al., 2017; Mia et al., 2018; Sanchez--Monedero et al., 2018), as a material limiting the emissions of greenhouse gases or odors (Janczak et al., 2017; Maurer et al., 2017) often accompanying the composting process, or as an additive influencing the improvement of the properties of the soil (Agegnehu et al., 2017; Czekała, 2019), the reduction the bioavailability of pollutants (Oleszczuk et al., 2012), plant growth and yield (Czekała et al., 2019; Ferreira et al., 2019) and heavy metal mobility (Uchimiya et al., 2011; Xue et al., 2012; Zama et al., 2017) and their accumulation in plant biomass (Zhao et al., 2016).

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Chapter 6

Improvement of biowaste management by identifying and promoting the best available technologies and practices

Rafał NOWAK

6.1. INTRODUCTION

Demographic growth, a consumptive lifestyle, and a high level of income increase the amount of generated waste. Unfortunately, depositing of such waste in disposal sites is still one of the main methods of their management. In the piles of accumulated waste, biodegradation of organic matter occurs. As a result, biogas is generated, whose main components are CH_4 and CO_2. The gas migration from dumping sites contributes to the deepening of the greenhouse effect. These factors have a global dimension and call for a change in approach to waste management. This is also facilitated by the lack of free space in landfills and no approval of local communities for the construction of new waste treatment plants or waste incineration plants.

Overexploitation of the environment, limited access to natural resources, and emission of greenhouse gases cause an increase in the importance of LCA (Life Cycle Assessment) or BAT (Best Available Technologies) in production and management processes. LCA is the method regulated with ISO 1404x standards (Güereca, 2019). Implementation of LCA methodology has unambiguously contributed to a broadened awareness of the product or technology effect from the perspective of an entire lifetime to environmental aspects. Interpretation of LCA results is not a simple task, however using it makes it possible to understand and evaluate the effect of the product/technology on the environment, but also to develop alternatives taking into consideration the scientific approach. LCA is an essential tool to make decisions connected with decreasing the negative impact on the natural environment.

The best available technology is a standard in the European Union, according to the IPPC Directive (Wiśniowska, 2015). The objective of bringing the IPPC Directive was to prevent pollution of the environment, control of the impact of non-preventing impacts as well as minimizing the effects on pollutants of individual components of the environment and the environment as a whole. The use of BAT

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refers to economic entities, whose business activity is a subject to the IPPC permit (integrated permit) and concerns pollutants generated in significant quantities. The IPPC Directive imposes an obligation to obtain an integrated permit, which determines the possibility of taking and running the selected economic activities (appendix I of IPPC directive). It is also necessary to use BAT – using these techniques is a condition to obtain an integrated permit. It also leads to the optimization of activities in the protection of the environment as a whole with simultaneous resignation from the technology that allows for the protection of only selected parts of the environment.

In the context of these two problems, sustainable waste management should be of the highest importance, with simultaneous reduction of methane emissions from dumping places.

6.2. WHAT IS THE DIFFERENCE BETWEEN WASTE AND BIOWASTE?

The European Parliament in Directive 2008/98/EC defined (OJL, 2008):

a) **waste** as: “any substance or object which the holder discards or intends or is required to discard”;
b) **biowaste** as: “biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants and other waste with similar biodegradability and compostability properties”.

According to the data given by Central Statistical Office (CSO) in Poland in 2018, about 128 mln tonnes of waste was generated, including 12 mln tonnes (9.8%) of municipal wastes (CSO, 2019). Per capita the quantity of generated wastes in Poland is equal to 325 kg and is lower than average in the European Union (486 kg). Yet despite these facts, in Poland the rates of the wastes that are treated are low, only 26% of wastes are recycled, 23% are thermally treated, and 8% are composted, or digested. Within the European Union, wastes are recycled at average of 30, 29% are incinerated, 23% are dumped, and about 17% are composted (CSO, 2019). It is generally thought that in mixed municipal waste about 45-47% are biodegradable, and 35% of them are biowastes (Majoch and Jabłońska, 2013). Figure 6.1 presents the quantity of municipal waste in Poland which is composted or digested. The map was drawn with DataWrapper, an open-source map-creator software.

Concerning the upwards trend in waste generation per capita, improvement in waste management via identification and promoting the best available techniques is a prerequisite for limiting methane emissions to the atmosphere, whereby sustainable waste management and implementing no- or low-waste technologies should be the overall objective of all member countries of the European Union.

The main threat to the environment resulting from uncontrolled dumping of wastes is methane gas (which is greenhouse gas), that is generated as a result of biological decomposition of organic matter under anaerobic conditions (Interreg Europe, 2017). Moreover, inappropriately landfilled biowastes can also contribute to eutrophication of the watercourses. In this context wrong waste management negatively affects the environment, and simultaneously potential resources are wasted. These resources can be e.g. incinerated (it allows for use of the energy from wastes). The energy can be used e.g. to generate electricity, in central heating, or in the heating of sanitary water.

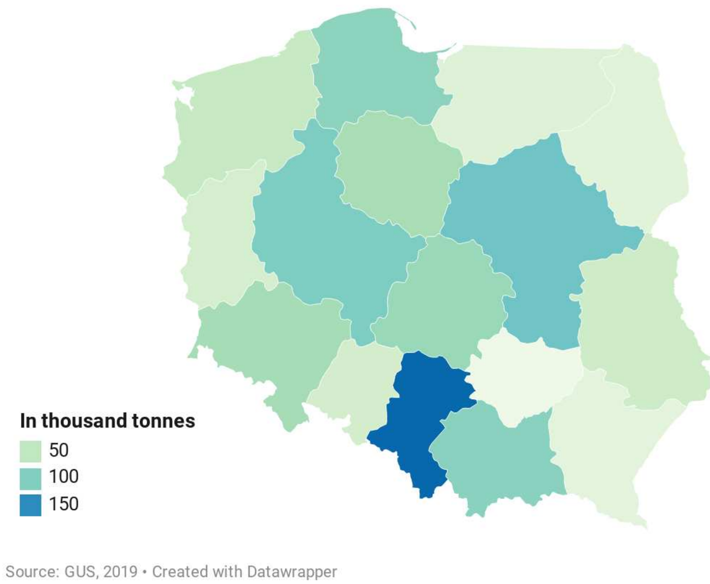

Fig. 6.1. The quantities of municipal waste in Poland which is composted or digested (CSO, 2019)

The European Union places great emphasis on sustainable development politics, both in the sphere of production or consumption or utilization. The example is Commission Implementing Decision (EU) 2018/1147 of 10 August 2018 establishing best available techniques (BAT) conclusions for waste treatment, under Directive 2010/75/EU of the European Parliament and the Council (OJ EU, 2018).

6.3. COMPOSTING AND BIODEGRADATION IN ENERGY PRISMS

The most frequently used method of biowaste utilization is composting. The raw material for this process are food residues, organic wastes from marketplaces, biowastes originating from servicing of urban greenery as well as stabilized sewage sludge. Composting of these materials allows for stabilizing and higienisation of organic wastes. This process also kills pathogenic organisms. As a result, the compost generated from the stabilization process, once when the rigoristic criteria concerning sanitary conditions and concentration of heavy metals are fulfilled, can be used as a fertilizer. Limit values of heavy metals in composts depend on how the compost will be used (e.g. recultivation of dumping places, industrial areas, arable soils) as well as on the concentrations of heavy metals in soil on which the compost will be used. The advantages of the composting process are generating the compost, higienisation of wastes, decreasing the volume of dumped wasted by 30-50%, simple technology, low investment and operational costs.

If biowaste contains a high concentration of the pollutants mentioned above and utilization of compost for the recultivation or in agriculture is impossible, the wastes can be degraded by use of energy prisms. To use this technology, separation of organic wastes from the municipal waste stream is necessary because wet organic wastes should not be landfilled. Organic wastes are sieved to obtain fine and average-size factions, followed by controlled mineralization. Biogas generated in the process contains 30-70% of CH_4, 30-60% of CO_2, dust, water vapor, sulfides, chlorides, nitrogen compounds, and hydrogen. The biogas is collected in wells and used for:

- Heat production – the burning of the biogas allows for the generation of heat energy. The biogas has to be cleaned from solid fraction, chlorides, sulfides, and water vapor and is possible when methane share in biogas is higher than 35%. The disadvantage of the process is the necessity of close localization of the receivers of the generated heat.
- Electricity generation – biogas is burned in motor generators and/or gas turbines (when methane percent share is higher than 30%).

The direct generation of energy from the biogas of low methane content is not economically justified. In this case supporting the burning process by using another fuel (e.g. natural gas) is necessary. What's more, burning of the biogas in torches (without energy recovery) is possible. If stored biogas is clean and caloric (its parameters are similar to natural gas), it is possible to supply it directly to the gas networks. The average energy value of the biogas generated from biowaste is equal to 21 MJ/m^3.

This method of biowaste management not only meets LCA philosophy, but also BAT. It is also an example of electricity generation from renewable sources within the meaning of the 2009/28/EC Directive (OJ EU, 2009). The advantage of the method is also that the waste generated during the process is an inert material, meaning that no pollutants are leached from the waste residue, and it can be e.g. used as material for the construction of recultivation layers (Ardolino et al., 2018; EESI, 2020; Klimiuk et al., 2012; Lin et al., 2018). In Figure 6.2, the possibilities of biogas utilization depending on the methane percent share are presented (Czurejno, 2005).

Obtaining biogas from biowaste can also reduce the import of natural gas and oil, while reducing greenhouse gas emissions. Processed biowaste is part of the circulation/recycling of nutrients, reducing the need for synthetic fertilizers (EESI, 2020). Composting and fermentation have advantages and disadvantages, so comparison of anaerobic and aerobic waste biostabilization was present in Table 6.1.

Table 6.1

Comparison of anaerobic and aerobic waste biostabilization (Jędrczak, 2018)

Criterion	**Methane fermentation / anaerobic stabilization**	**Composting / oxygen stabilization**
Technology development	in a state of development	state of the technology
Raw materials	2 ingredients (biowaste + water) and heat	3 ingredients (biowaste + water + air), possibly structural material
Products	– biologically stabilized digestate (required dehydration and oxygen stabilization) – biogas (high-energy gas) – sewage (treatment required)	– compost (for sale) or stabilizer – post-process air required purification on biofilters – condensates, sewage (recirculation recommended, excess – cleaning)
Environment: – oxygen – optimal substrate moisture – nutrients – pH value – temperature	– anaerobic process – from 60 to 90% – C/N = 10:1-30:1 – from 6.5 to 8.0 – 35°C (mesophilic process) – 55°C (thermophilic process)	– oxygen process, from 5 to 15% O_2 in the air in the pores – from 40 to 60% – C/N = 20:1-35:1 – from 5.5 to 8.0 – up to 60°C
Degree of decomposition of organic substances	from 45 to 67%	approx. 55%
Nature of the process	endothermic	exothermic
Energy demand	as a rule, excess energy	energy-intensive process (continuous aeration)
Sanitary properties of the product	sanitary product, only after thermophilic fermentation	sanitary product
Odour emission	– non-odour fermentation (process carried out in a hermetic installation) – emission in the process of acceptance and pre-treatment and confectioning of the product (recommended purification on biofilters)	– in all stages of the process (purification required on biofilters)
Sewage – Quantity, dm^3/Mg – COD, g/dm^3 – BOD_5, g/dm^3 – NH_4^+, mg N/dm^3	– 200-350 – 0.50-2.5 – 0.10-1.2 – 15-300	– 10-60 (leachate) – 10-100 – 5-45 – 50-800
Process duration (weeks)	– process: 2-3 – treatment after the process: oxygen stabilization; 2-8	– process: 18-16 – treatment after the process: –
Individual space requirement	**from 0.2 to 0.4 m^2/Mg**	**from 0.3 to 0.6 m^2/Mg**

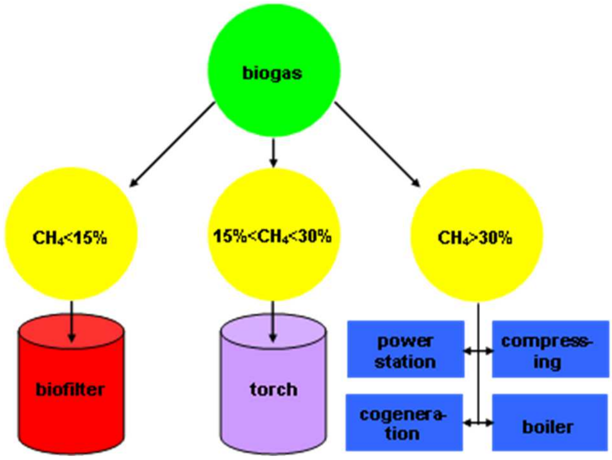

Fig. 6.2. Possibilities of biogas utilization depending on the percent share of methane (Czurejno, 2005)

The choice between both of the biowaste processing methods described above is determined by local conditions and applicable law regulations.

6.4. BIOWASTE AND LEGAL REGULATIONS

Framework Directive 2008/98/EC imposes an obligation on the Member States to selectively collect biowaste and its subsequent processing while ensuring appropriate environmental standards and then using the resulting waste in an environmentally safe manner. In Polish legislation, the issue of biowaste is regulated by:

- Act of 14 December 2012 on waste (Journal of Laws, 2020);
- Act of 19 July 2019 amending the act on maintaining cleanliness and order in municipalities and certain other acts (Journal of Laws, 2019).

According to these provisions, biowaste is considered to be quoted as: "biodegradable garden and park waste, food and kitchen waste from households, gastronomy, caterers, retail units, as well as comparable waste from plants producing or marketing food". This definition helps in the identification of this type of waste from the municipal waste stream and facilitates its subsequent management taking into account BAT.

The Waste Act also introduces the issue of loss of waste status (Chapter 5, Article 14) if, as a result of the recovery and recycling process, the following conditions are met: "a) the object or substance is commonly used for specific purposes; b) there is a market for or demand for such items or substances; c) the object or substance meets the technical requirements for use in specific purposes, and standards applicable to the product, d) the use of the object or substance does not lead to negative effects on human life, health or the environment". The waste loss status also applies when the "requirements set out in European Union legislation" are met (OJ, 2020). Article 15 of the same Act stipulates that the joint storage of waste and substances that no longer have the status of waste and their storage in places designated for waste storage is prohibited.

6.5. SUMMARY

The creation of a selective waste collection system, the introduction of which has been determined by current legislation, promotes the identification of biowaste from the municipal waste stream and is a key issue for the proper management of biowaste. The use of the best available technologies for this type of waste reduces greenhouse gas (GHG) emissions, biogas, and renewable energy production and helps meet the requirements for a 15% share of renewable energy sources in the total balance imposed by Directive 2009/28/EC consumed energy. The product remaining after the composting process, after meeting the appropriate conditions, can be used economically, and thus reduce the amount of landfilled waste. That is why it is so important to implement, both at the regional and international level, sustainable biowaste management and the use of BAT in strategic planning. In the context of the depletion of non-renewable resources, the increase in the costs of their extraction and the associated environmental pollution, obtaining energy from biowaste seems to be an informed choice, economically justified, and in line with the integrated waste management strategy.

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Chapter 7

Identification, quantification and prioritization of technological solutions from the point of view of environmental impact

Elżbieta SPERCZYŃSKA

7.1. INTRODUCTION

The method of LCA (Life Cycle Assessment) is a technique in the field of management processes intended to assess potential environmental hazards. According to the ISO Standard, LCA involves “compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle” (ISO 14040, 2009) and is an analytical tool that can be applied in many areas of environmental management. The essence of this method is the attitude to not only evaluate the final result of the manufacturing process but to also evaluate and assess the consequences of the whole process for the environment. It represents a compilation and evaluation of all inflows (inputs), effluent (outputs) and the potential environmental impact of a certain production system throughout its life cycle. Through such an overview, the shifting of a potential environmental burden between life cycle stages or individual processes can be identified. The LCA method is currently the only internationally standardized method for assessing the impacts of products throughout their life cycles.

Multinational corporations consider LCA as a tool to influence the decisions of numerous suppliers and consumers. LCA is considered as a methodological basis for decisions in terms of preferences for certain types of raw materials and auxiliary substances.

The LCA method’s objective is to (Lewandowska, 2006):

- quantify all relevant flows of raw materials consumed and pollutants emitted throughout the supply chain;
- comprehensively assess the potential impacts on the environment and human health of the entire supply chain of a product and identify hotspots of environmental impacts across the supply chain;

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- ➢ identify trade-offs between life-cycle stages, impact categories or regions that can lead to a shifting of environmental burdens.

Considering the life cycle of the product, as a balance between material and energy, particular attention should be given to a manufacturing system model due to the possibility of sustainability all production factors from the production system planning model (Hur et al., 2005; Khasreen et al., 2009).

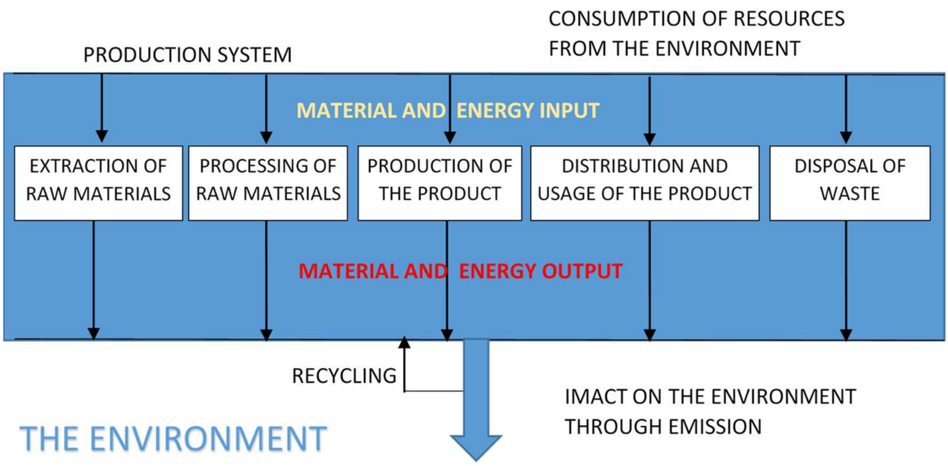

Fig. 7.1. Life Cycle of product (ISO 14040)

This method of analysis aims to gain insight into the entire product life cycle (Fig. 7.1), which includes:

- ➢ extraction of raw materials,
- ➢ the acquisition of energy resources,
- ➢ production and distribution of energy required,
- ➢ production of semi-finished products and by-products,
- ➢ transportation and distribution,
- ➢ effects during use,
- ➢ alternatives handling of the product after use.

Such an approach is particularly important when there are alternative routes and you can choose a version that is less harmful to the environment. LCA analysis allows you to understand the input data and output products at all stages of production (Grimauda et al., 2018).

7.2. PRINCIPLES OF LCA RESEARCH

The LCA study range is carried out in four phases outlined in the Standard ISO (ISO 14040:2009; ISO 14044:2009):

Phase 1 – to determine the purpose and scope of the analysis.

Phase 2 (Life Cycle Inventory – **LCI**) – analysis of the sets of inputs and outputs (analysis of the technological process, material and energy required for the process and emissions and waste, as well as the identification of potential sources of their formation considering issues of intangible assets, such as noise and odor – Fig. 7.2).

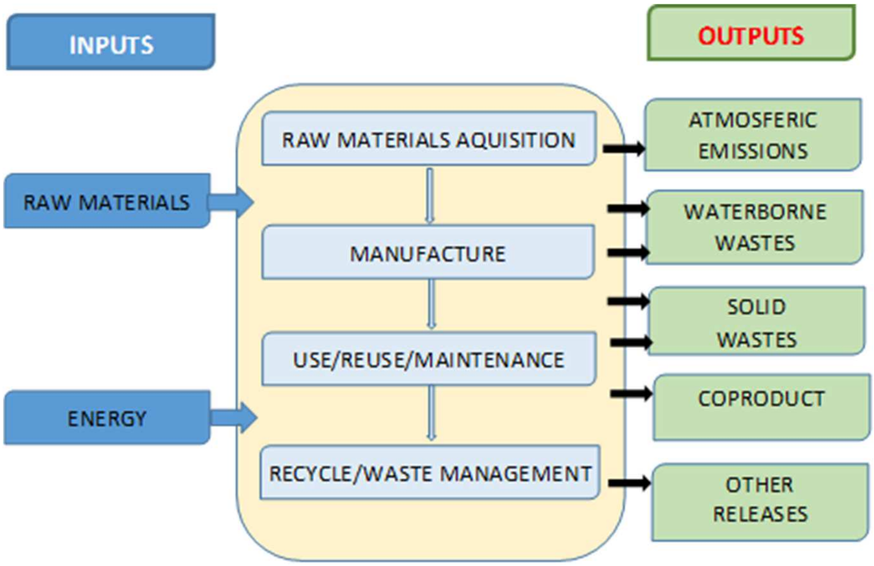

Fig. 7.2. Life Cycle stage (Perić et al., 2016)

Phase 3 (Life Cycle Impact Assessment – **LCIA**) – life cycle impact assessment on the environment (transformation of the data collected in the impact category indicators or categories of damage).
Phase 4 – results interpretation (application and verification of results).

The procedure for performing the life cycle assessment is shown in Figure 7.3. The first step is to define the purpose and scope of the study. The purpose and scope of the study should specify the intended use, the reasons for the study and the recipient of the results. The purpose of the tests determines the level of accuracy of the LCA, and the scope of tests describes individual processes, its limits and functional unit. The smallest unit accepted for research is the functional unit, which becomes the quantitative effect of the LCA system. The main task of a functional unit is to provide a reference plane for the standardization of input and output data of a specific system, therefore it should be clearly defined and measured.

The simplest functional units are physical units such as meters, joules, kg, seconds, Kelvin, etc. Single units can be combined into complex ones, e.g. tonne-kilometer (for transport), m^2 year, lumen year, etc. For example, in the Raghuvanshi et al. (2018) study, the functional unit is 1 MJ of energy produced from biodiesel from fresh water and wastewater. Functional units can also be a single device/machine or one of the functions of this device or a development area for which the flow is determined as materials and energy. The choice of the functional unit can strongly affect the impact results of LCA studies (Zhao and Pedersen, 2018).

BEGINNING

Phase 1 **Defining aim and scope**	**1. Objective of the action** **2. Defining the choice** **3. Choosing system limitations** **4. Choosing environmental parameters** **5. Choosing methods of gathering and assessing data** **6. Strategy of data gathering** **7. Initial initiation of LCA method** **8. Verifying the step of 'defining aim and scope'**
Phase 2 **Set analysis**	**1. Measurements, theoretical calculations** **2. Characteristics of processes used** **3. Defining product-in and product-out of the process** **4. Assigning data tables of stocktaking** **5. Classifying impact on environmental impacts**
Phase 3 **Assessment of impact**	**1. Allocating data to impacts** **2. Defining a catalogue of wastes** **3. Classifying data tables of stocktaking in terms of kind of impact** **4. Characteristics of particular classes** **5. Standardization of results** **6. Hierarchizing classes (evaluation of impact))**
Phase 4 **Interpretation**	**1. Defining elements of high risk** **2. Sensitivity analysis of above** **3. Determining manner of minimizing risks** **4. Assessing priorities of possible corrections and their feasibility**

END

Fig. 7.3. LCA implementation procedure (Dzikuć and Piwowar 2013; Wach, 2002)

The second phase (LCI) is a process of quantifying energy and raw material requirements, atmospheric emissions, waterborne emissions, solid wastes, and other releases for the entire life cycle of a product, process, or activity. In the LCI phase of an LCA, all relevant data is collected and organized. Without LCI, there is no basis for assessing comparative environmental impacts or potential improvements. The level of accuracy and detail of the data collected is reflected in the rest of the LCA process.

Steps of a life cycle inventory:

1. Develop a flow diagram of the processes being evaluated.
2. Develop a data collection plan.
3. Collect data.
4. Evaluate and report results

Each step in the life cycle of a product, package, or material can be categorized within one and only one of these life-cycle stages. Each step or process can be analyzed as a subsystem of the total product system. Examination steps as subsystems facilitate data collection for the inventory system as a whole.

In LCI, the results are presented as tables and graphical studies. Life-cycle inventory studies generate a great deal of information, often of a disparate nature. The received data should be grouped, e.g. (US EPA, 2006):

- Overall product system,
- Relative contribution of stages to the overall system,
- Relative contribution of product components to the overall system,
- Data categories within and across stages, e.g., resource use, energy consumption, and environmental releases,
- Data parameter groups within a category, e.g., air emissions, waterborne wastes, and solid waste types,
- Data parameters within a group, e.g., sulfur oxides, carbon dioxide,
- Geographic regionalization if relevant to the study, e.g., national versus global.

The Life Cycle Impact Assessment (LCIA) phase of an LCA is the evaluation of potential human health and environmental impacts of the environmental resources and releases identified during the LCI. A life cycle impact assessment attempts to establish a linkage between the product/process and its potential environmental impacts and human health effects.

Based on the LCI results, the category of midpoints and their environmental impact are determined. In particular, they indicate damage caused by assessed parameters. Midpoint categories (Jolliet, 2004): Human toxicity, Respiratory effects, Ionizing radiation, Ozone layer depletion, Photochemical oxidation, Aquatic ecotoxicity, Terrestrial ecotoxicity, Aquatic acidification, Aquatic eutrophication, Terrestrial acid/nutrient, Land occupation, Global warming, Non-renewable energy, Mineral extraction.

Steps of a Life Cycle Impact Assessment (US EPA, 2006):

1. Selection and Definition of Impact Categories – identifying relevant environmental impact categories (e.g., global warming).
2. Classification – assigning LCI results to the impact categories (e.g., classifying carbon dioxide emissions to global warming).
3. Characterization – modeling LCI impacts within impact categories using science-based conversion factors (e.g., modeling the potential impact of carbon dioxide and methane on global warming).

4. Normalization – expressing potential impacts in ways that can be compared (e.g. comparing the global warming impact of carbon dioxide and methane for the two options).
5. Grouping – sorting or ranking the indicators (local, regional, and global).
6. Weighting – emphasizing the most important potential impacts.
7. Evaluating and Reporting LCIA Results.

Interpretation is the phase of LCA in which the findings from the inventory analysis and the impact assessment are considered together or, in the case of LCI studies, the findings of the inventory analysis only. Moreover, this phase makes it possible to define elements of great risk, to define the manner of minimization of threats as well as to assess priorities of possible corrections and their feasibility.

The interpretation phase should deliver results that are consistent with the defined goal and scope, and which reach conclusions, explain limitations, and provide recommendations. The assessment conducted according to the mentioned procedure allows one to present the results of impact in relation to impact categories included in Table 7.1.

Table 7.1

Categories of environmental impact

No	Category	Description
1	Abiotic impoverishment	Extraction of non-renewable ores of mineral resources
2	Energy impoverishment	Extraction of non-renewable energy carriers. This category can be included in category 1
3	Greenhouse effect	Atmospheric absorption of radiation leading to the increase of global temperaturę
4	Ozone hole	Increase of ultraviolet radiation reaching the surface of Earth caused by impoverishment of ozone layer
5	Water and soil contamination	Exposing biota to toxic substances
6	Acidifi cation	Increase of water and soil acidity
7	Contamination of humans	Exposing human health to toxic substances appearing in water, air and soil, mainly with food
8	Creating photochemical oxidants	Emergence of atmospheric particles causing photochemical smog
9	Eutrophication	Reduction of oxygen amount in water or soil by emission of substances causing increase of biomass production

ISO standards for all phases of analysis recommend the following support activities in the relevant product system (Dzikuć and Piwowar, 2013):

- preparing detailed diagrams of processes' transfers together with existing individual processes and relations between them,
- elaborating a description of all individual processes together with a list of the categories of data related to particular processes,

- preparing a description of methods for gathering data and methods of calculating,
- elaborating instructions regarding the places of gathering data,
- elaborating a catalogue which includes measuring units (ISO 14044:2009).

7.3. INDICATORS IN LCA RESEARCH

Essential indicators that are required to be determined in Life Cycle Assessment (Gray and Bebbington 2001; Ingaldi et al., 2016):

- **Raw materials**: specification and indication of the amount of raw materials used in kg.
- **Water**: water consumption is attributed to the final product produced.
- **Transport**: considers the payload of the transport and the length of the road it takes.
- **Energy consumption**: the energy consumption of all phases of the production process.
- **Emissions to the atmosphere**: emissions to the atmosphere in all phases of the production process and arising during transport.
- **Waste**: different types of waste allocated to the respective production phases of the process.

The environmental indicators are determined by three groups of parameters (Ingaldi et al., 2016):

a) Resources consumption

- consumption of non-renewable resources, kg, MJ,
- consumption of renewable resources, kg, MJ,
- consumption of primary energy sources, MJ,
- consumption of electricity, kWh,
- consumption of water, m^3.

b) Pollutant emissions expressed by the potential impact on the environment

- global warming, kg eq CO_2,
- acidification, kmol eq H^+,
- destruction of the ozone layer, kg eq CFC^{-11},
- the formation of photochemical oxidants, kg eq C_2H_4,
- eutrophication, kg O_2.

c) The generated waste

- hazardous waste, kg,
- waste for recycling, kg,
- other waste, kg,
- depending on the type of materials used, information about the emissions of the following substances must be defined: SO_2, NO_x, Cd, Cr, Hg, Ni, Pb, Zn.

Impact indicators are typically characterized using the following equation (US EPA, 2006):

Inventory Data × Characterization Factor = Impact Indicators

Characterizing factors (CF) convert different inventory inputs into directly comparable impact indicators. Characterization factors (also called equivalence factors) are scientifically calculated conversion factors that are used to convert the assigned LCI results to a common unit of a category indicator. The characterization factors are determined for many different substances associated with different impact categories in the life cycle assessment (Alfonsín et al., 2014; US EPA, 2006).

For example, pollutant emissions expressed by the potential impact on global warming can be expressed in terms of CO_2 equivalents. To obtain a global indicator of global warming potential (GWP), multiply the relevant LCI results by the CO_2 characterization factor, and then combine the resulting impact indicators.

This method allows to put different amounts of chemicals on a uniform scale to determine the impact of each of them on global warming.

For example: if we have the following Inventory Data (US EPA, 2006):

Chloroform – 20 kg
Methane – 10 kg

Characterization Factor Value for Chloroform = 9
Characterization Factor Value for Methane = 21
Characterization Factor according for Intergovernmental Panel on Climate Change (IPCC) Model

Chloroform GWP Impact = 20 kg x 9 = 180 kg eq CO_2
Methane GWP Impact = 10 kg x 21 = 210 kg eq CO_2

The calculations show that ten kilos of methane have a larger impact on global warming than twenty kilos of chloroform.

7.4. CONCLUSIONS

Life Cycle Assessment is the most advanced formal and legal environmental management tool recommended by European law. It is a universal method and can be used to determine the environmental impacts of both products and processes in many different service and industrial industries.

Based on the results of LCA analysis, it is possible to determine the friendliest product production system that is the most beneficial / least harmful to the environment. The product impact analysis starts with the product design stage, which is to guarantee objectivity and determine all impacts. This stage is a critical point for any project or product because it implies factors determining the degree of use of natural resources.

A very important tool in the LCA analysis is the Life Cycle Impact Assessment (LCIA), which assesses the potential impact on human health of the environment. Defining the category of midpoints allows you to choose the most environmentally friendly technology. LCA is the ideal tool for comparing the environmental impacts of competing products and identifying key areas where improvements could be made.

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Chapter 8

Identification of factors that may influence bioprocessing technology in the future

Iwona DESKA

8.1. INTRODUCTION

Waste generation is one of the major problems for our society in the 21st century (Coma et al., 2017). Biowaste constitutes a significant fraction of municipal solid waste (MSW) (Pavlas et al., 2020). Biowaste is the predominant waste fraction in low- and middle-income settings (Lohri et al., 2017) (De Morais Vieira and Matheus, 2019). A large amount of agro-industrial waste is generated each year which leads to economic loss and environmental pollution (Fierascu et al., 2019). About 50% of global waste (3 million tons per day) is organic waste. For example, about 700 million tons of agricultural waste is annually generated within old European Union countries (EU) (Fritsch et al., 2017). It includes, among others, household waste, food manufacturing, and pre-factory waste. It also contains certain amounts of paper, plastic, glass and metal etc. (Coma et al., 2017). Every year, about one third of globally produced food (approximately 1.3 billion tons) is lost or wasted. Food loss is a decrease in the quantity or quality of food. It results from decisions and actions by food suppliers in the chain. Food waste, in turn, is a decrease in the quantity or quality of food and results from decisions and actions by retailers, food service providers and consumers. Less food loss equals more efficient land use as well as better water resource management (FAO, 2011). Food loss as well as food waste are huge economic and environmental problems that will continue to rise in the future.

Organic waste is one of the major sources of greenhouse gases (GHG) but, on the other hand, it contains different compounds, most of which have both energetic and economic value (Coma et al., 2017). Treatment of biowaste offers economic, environmental, and public health benefits enabling the conversion of waste into hygienic products. Biowaste treatment technologies can be grouped into four main categories: direct use, biological treatment, physico-chemical treatment, and thermo-chemical treatment (Lohri et al., 2017).

The biological treatment process (or bioprocess) is the controlled conversion of waste by living organisms that takes place in a moist environment.

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The bioprocesses used for the biowaste treatment include among others: anaerobic digestion (AD), composting, vermicomposting, fermentation (e.g. ethanol and dark fermentation), and Black Soldier Fly (BSF) treatment (Lohri et al., 2017; De Morais Vieira and Matheus, 2019).

The aim of this chapter is to characterize the selected bioprocesses used for biowaste treatment with a particular reference to the parameters and conditions that influence these processes. The chapter is also devoted to the advantages and limitations of described selected bioprocesses as well as their applications. The primary focus of the chapter is the identification of the factors that may have an impact on bioprocessing technology in the future.

8.2. THE FACTORS INFLUENCING MAJOR BIOPROCESSES

8.2.1. ANAEROBIC DIGESTION (AD)

Anaerobic digestion (AD) (also known as biomethanization or biomethanation) is a well-established engineered process in which organic matter (liquid and solid) is biochemically decomposed by various bacterial activities in an oxygen-free environment. AD occurs naturally in different anoxic environments, for example in soils, landfills, surface water as well as in animal intestines (Lohri et al., 2017; Vögeli et al., 2014). The conversion of food waste into energy (e.g. methane CH_4) via anaerobic processes is economically viable. The anaerobic biodegradation that occurs during AD consists of the following four microbial processes as: hydrolysis, acidogenesis, acetogenesis, and methanogenesis (Paritosh et al., 2017). In this process, methanogenic organisms and devices such as hydrolisers and fermenters allow the oxidation of variable substrates to a single product (Coma et al., 2017).

The key parameters of AD are: temperature, pH, moisture, substrate carbon to nitrogen ratio (C:N ratio), organic loading rate (OLR), hydraulic retention time (HRT), inoculation and start-up, mixing, and inhibition (Lohri et al., 2017; Vögeli et al., 2014).

Methane production is largely dependent on the temperature, the majority of AD systems operate effectively at mesophilic conditions. The temperature decrease leads to the reduction of bacterial activity resulting in turn in the decrease of CH_4 production (Anukam et al., 2019). The temperature decides the inhibition/ stimulation of a particular microorganism kind. The optimal temperature for the survival of thermophilic bacteria is 55°C, but the optimal temperature for the survival of mesophilic bacteria is 35°C (Singh et al., 2017). In the case of a mesophilic digester, there are lower heater energy costs compared to the thermophilic one. But, on the other hand, mesophilic digestion is slower and yields less biogas (Meegoda et al., 2018; Wang et al. 2019). The optimum pH for a stable AD and high biogas yield ranges between 6.5 and 7.5. Hydrolysis and acidogenesis take place at acidic pH conditions (from 5.5 to 6.5) while methanogenesis takes place at a pH which extends from 6.5 to 8.2). Substances like: lime, sodium bicarbonate or sodium hydroxide can be used for pH adjustment. The digester should be inoculated with bacteria necessary for the anaerobic process. The good inoculate can be e.g. cow dung (1:1 ratio with water) and digested sludge from a sewage plant. The population of bacteria should be gradually acclimatized to the feedstock during the start-up phase of AD which can be achieved by a gradual rise of the daily feeding load (Vögeli et al., 2014). Food waste degradability is strongly dependent upon its chemical composition. It is very important to know the percentages of different components of the heterogeneous substrates used in the AD process (Paritosh et al., 2017).

Table 8.1

Advantages and limitations of anaerobic digestion (Paritosh et al., 2017; Vögeli et al., 2014)

Advantages
– Biogas produced during AD can be used for heat production as an alternative for fossil fuels, e.g. wood, LPG, charcoal. – Biogas produced during AD can be stored in a gastight container for long periods of time without losing its energy content. – Biogas production can provide the additional incomes. – AD provides for recovery of resources and conservation of the renewable energy sources. – The biogas production using AD technology is feasible under almost all climatic conditions. – AD is a method of the feedstock waste sanitization. – The effect of the fertilizer produced during AD process is longer lasting than the effect of untreated waste. – The solid state AD requires smaller volume of reactor than the liquid AD. – AD can reduce odour below the odour levels of unprocessed waste. – Lower mass of the generated sludge compared to an aerobic system.
Limitations
– The high investment and operating costs. – The possible smell of the effluent. – Not enough biogas in the case of small installations. – The not satisfactory work under the relatively low mean temperatures (below 15°C). – The large temperature variations (e.g. seasonal variations) may have unfavorable impact on the AD efficiency. – The relatively low reaction rate in the case of solid state AD. – The longer duration of initial stage of process compared to aerobic systems. – AD can have high requirements of buffer chemicals to keep the proper range of pH during process.

The C:N ratio is a very important organic material parameter whose optimal value in anaerobic digesters ranges between 16 and 25. The high C:N ratio leads to lower gas production due to the rapid consumption of nitrogen by methanogens. The low C:N ratio, on the other hand, contributes to an increase of pH values (pH > 8.5) due to the accumulation of ammonia which may be toxic for methanogenic bacteria (Vögeli et al., 2014). The OLR is the quantity of substrate that is introduced into the specified volume of the reactor in a given time. The OLR increase contributes to the rise in volatile fatty acids and acidification of the AD system. The HRT is the time during which liquid remains in the reactor. In the case of the mesophilic digester, the optimal HRT depends on the technology type and ranges from 10 to 40 days. The mixing and stirring prevent temperature gradients within the digester and also prevent scum formation. Inhibition a very important factor affecting AD process which depends on both the substrate composition and the concentration of inhibitors such as O_2, H_2, H_2S, organic acids, free ammonia, heavy metals, antibiotics etc. It is very important to avoid acidification and inhibition of methanogenic bacteria during AD (Vögeli et al., 2014). Table 8.1 presents the advantages and limitations of AD.

8.2.2. COMPOSTING

Composting comprises the controlled aerobic organic matter decomposition that results in the relatively stable organic end product called humus. (Hanc and Pliva, 2013; Schmidt et al., 2018). During this process, microbial respiration leads mostly to emission of carbon dioxide (CO_2) while the mineralization of nitrogen (N) results in an ammonium (NH_4^+) production (Vergara and Silver, 2019). Composting is a prevailing process of biowaste degradation in Europe (Schmidt et al., 2018). This is one of the most sustainable methods of handling food waste and includes three phases: the decomposition (pre-composting), the transitional stage, and curing stage.

Composting can be conducted in open systems (turned and static pillows and turned windrows) and closed systems (reactors, rotating drums and tanks) (Hanc and Pliva, 2013; Jędrczak et al., 2018). On the other hand, a relatively small part of biowaste is digested for biogas production. The disadvantages of composting are the high loss of energy in the form of heat and the release of carbon dioxide (Schmidt et al., 2018). The parameters influencing the composting process carried out in the open air are the moisture content, oxygen, C:N ratio and waste air porosity. Composting carried out in the reactors allows for better control of the process parameters (e.g. oxygen content, humidity and the temperature) as well as control of emissions to the environment due to the possibility of capturing and purifying the contaminated air. The choice of the composting system is dependent on the following local conditions: the type of waste, the space availability, and the installation capacity (Jędrczak et al., 2018). Table 8.2 presents the advantages and limitations of composting.

Table 8.2

Advantages and limitations of composting (Vergara and Silver, 2019; Schmidt et al., 2018)

Advantages
– The greenhouse gas benefit due to the elimination of anaerobic storage which is the greenhouse gas benefit. – The composting of biowaste can generate the revenues. – Lower initial capital investment compared to the AD plant. – The regulation and increase of water storage in soils due to the compost addition. – Addition of the compost has a positive impact on a permeability of clay soils for both water and air. – Due to the slow release of nitrogen in compost this macroelement can be available to plants during a relatively long period of time.
Limitations and challenges
– Relatively high energy costs due to the aeration and turn of the compost piles. – The release of carbon dioxide and high loss of energy in the form of heat. – Composting makes no contribution to reducing the carbon footprints of the businesses that use the composting process. – The essential problem connected with composting is the control of odors, especially if the process runs inefficiently. – Composting is strongly influenced by the weather conditions.

8.2.3. VERMICOMPOSTING

Vermicomposting is an aerobic process of organic waste degradation (bio-oxidation) and stabilization. This bioprocess can be considered as the advanced composting technique that uses earthworms. Vermicomposting is influenced by the interaction between microorganisms and earthworms, takes place under controlled conditions and comprises diverse techniques, e.g.: small-scale domestic systems (containers), windrow and batch systems, and the continuous flow vermicomposting reactors (Hanc and Pliva, 2013; Lohri et al., 2017).

Through a first step of aerobic degradation, the microorganisms prepare the waste for earthworms. Earthworms are able to process both household and organic municipal waste. They can process organic waste residues from different industries and sewage sludge. But, on the other hand, earthworms do not tolerate meat and fish waste, dairy products, oils etc. Vermicompost has higher levels of nutrients than compost (Lohri et al., 2017).

Vermicompost is beneficial to germination, growth as well as the yield of plants. It improves the physical and chemical properties of soil. Vermicompost has a positive impact on soil health because its addition leads to an increase in both the organic matter content and water holding capacity. Vermicompost contains high concentrations of plant-available nutrients (e.g. phosphates, nitrates, soluble potassium, exchangeable calcium). Earthworms excrete the proteins and polysaccharides, while nitrogenous compounds enhance the microbial population in the soil. When the application of vermicomposting is excessive (the nutrients are supplied in excess), plant growth can be adversely affected. This can lead to high salinity and elevated concentrations of heavy metals that inhibit plant growth. Elevated salinity slows down water uptake by seeds. The high levels of phenolic and humic substances may suppress plant growth. For example, they increase the permeability of cell membranes and can lead to the lipid peroxidation. The phenolic compounds have an adverse influence on photosynthesis because they reduce the chlorophyll content, reduce the RNA and DNA integrity and hinder the activity of hormones (Hussain and Abbasi, 2018). Table 8.3 presents the advantages and limitations of vermicomposting.

Table 8.3

Advantages and limitations of vermicomposting (Hussain and Abbasi, 2018; Lohri et al., 2017; Kokhia, 2015)

Advantages
– Vermicompost has the same advantages than compost. – Vermicompost has higher levels of nutrients than compost. – Vermicompost exceeds 4-8 times manure and compost in the content of humus. – Vermicompost contains humic acids and all the necessary nutrients for plants. – During vermicomposting most human pathogens are killed by the action of soil microorganisms and digestive enzymes of worms.
Limitations and challenges
– The excessive application of vermicompost may inhibit the plant growth due to the high salinity and elevated concentrations of heavy metals. – The phenolic substances present in the vermicompost may adversely influence the photosynthesis due to reduction of the DNA and RNA integrity and lowering of the chlorophyll content. – During the production of vermicompost can be developed the potentially pathogenic mold and other pathogens.

8.2.4. ETHANOL FERMENTATION

Ethanol fermentation (also known alcoholic fermentation) is the key process step in bioethanol production. Bioethanol (ethyl alcohol, CH_3CH_2OH or EtOH) is the leading environmental-friendly and clean-burning biofuel. Bioethanol is produced from sugar-, starch-, and lignocellulose-based biomass using different conversion technologies. The essential materials used for production of bioethanol are corn-derived feedstock (starch) and sugarcane-derived feedstock (saccharose) (Lohri et al., 2017). There are three major groups of the carbohydrate sources for production of bioethanol: (a) simple sugars from sucrose containing feedstocks (sugarcane, sugar beet, sweet sorghum, molasses and fruits), (b) starchy materials, e.g. grains (corn, wheat, barley, rice), and root crops (potato, cassava), (c) lignocellulosic biomass (woody materials, straw, agricultural waste and crop residues) (Balat and Balat, 2009; Bušić et al., 2018).

The availability of raw materials is one the of major factors affecting bioethanol production. The availability of these materials varies considerably throughout the year and is dependent on the geographical location. Additionally, the prices of the feedstocks are highly volatile (Balat and Balat, 2009). The simple sugars (about 40% of the global bioethanol production) and starchy materials (about 60%) are classified as 1st generation, edible feedstock, and the lignocellulosic biomass is classified as 2nd generation, non-edible feedstock. Non-edible lignocellulosic feedstocks are suggested as the main materials for sustainable bioethanol production in the future (Lohri et al., 2017; Vohra et al., 2014). The disadvantage and the barrier to the production of second-generation bioethanol (from lignocellulosic biomass) is the difficult and costly pretreatment. The residues from the pulp and paper industry that are rich in monosaccharides (or even in polysaccharides) could be used as the material for bioethanol production. The use of lignocellulosic biomass (e.g. cellulose, lignin and hemicelluloses) to produce various products (among others bioethanol) can be crucial to both maximize biomass industrial processing and minimize generation of waste (Branco et al., 2019). Applying organic fraction of municipal solid waste as a feedstock in biorefineries can generate high revenues, especially in the case of production of biopesticides and enzymes. Organic fraction of municipal solid waste is an inexpensive material and should be considered as a resource (not as a type of waste) (Blikra Vea et al., 2018).

Table 8.4

Advantages and limitations of ethanol fermentation (Branco et al., 2019; Coma et al., 2017)

Advantages
– Bioethanol is easy applicable, renewable and biodegradable. – Combustion of bioethanol generates lower emissions (eg. of CO_2). – Bioethanol used in the engines has the advantages compared with gasoline. – An advantage of using bioethanol is its better combustion due to the higher oxygen content. – Bioethanol can be used as a platform chemical e.g. in beverages, pharmaceuticals, and cosmetics.
Limitations and challenges
– The utilization of the arable land in the case of bioethanol production from the first generation feedstock (the edible feedstock). – The possibility of an increase of food prices in the case of bioethanol production from the first generation feedstock. – The costs of the enzymatic hydrolysis are high. – The times of hydrolysis duration are long and reactions with use of enzymes are relatively slow.

The major contributor of food waste (53%) is the household. The disposal of food waste in landfills generates both greenhouse gas emissions and the formation of the leachate and closes the opportunity to use it as a feedstock. Household food waste can be a suitable source of carbon and microorganisms for production of volatile fatty acids. Household food waste should be considered a promising feedstock for the production of chemical intermediates like carboxylic acids and platform chemicals (Herrero-Garcia et al., 2018). Table 8.4 presents the advantages and limitations of ethanol fermentation.

8.2.5. DARK FERMENTATION (DF)

Fossil fuels are gradually being depleted, and hydrogen can replace them as a clean and sustainable alternative energy resource. Hydrogen is considered as the fuel of the future and has the ecological value (Kucharska et al., 2019). Hydrogen is a fuel that does not contribute to greenhouse gas emissions and, additionally, has very high energy efficiency because it produces 2.5 times more energy per kilogram than hydrocarbons. Hydrogen is a renewable energy and can be used as energy source for transportation (Moreno Cárdenas et al., 2019). Biohydrogen can be produced by fermentation (dark fermentation or photo fermentation) or photosynthesis (using direct or indirect photolysis) (Rathore et al., 2019). Dark fermentation (DF) is the process composed by the hydrolysis and acidogenesis – the first two stages of AD. Biohydrogen production is influenced by the type of substrate, pH, temperature, oxidation-reduction potential, addition of nutrients, or inhibitors. Wongthanate et al. (2014) found in their investigations that in the case of food waste and beverage processing wastewater, the hydrogen production by sewage microflora was optimized at low initial pH (6.5) and at mesophilic conditions (35 ± 2°C). They concluded that the generation of H_2 can be enhanced by pre-treatment methods such as sterilization, sonication, acidification, using of methanogenic inhibitors, freezing (thawing) of organic waste, etc. (Wongthanate et al., 2014). Table 8.5 presents the advantages and limitations of dark fermentation.

Table 8.5

Advantages and limitations of dark fermentation (Branco et al., 2019; Mishra et al., 2019; Tapia-Venegas et al., 2015)

Advantages
– The biohydrogen is considered as the carbon free fuel. The only byproduct after hydrogen combustion is water. – The hydrogen production rate during the dark fermentation can be order of magnitude larger than those achieved by other methods. – The possibility of an environmentally friendly production of hydrogen from renewable sources (eg. biomass). – The possibility of coupling the dark fermentation (hydrogen production) with AD. – The solid state fermentation for hydrogen production could be beneficial solution due to the lower water requirements and the use of smaller reactors.
Limitations and challenges
– The main challenges related to hydrogen production by dark fermentation are among others the stability of the processes and low conversion yields joined at the byproducts formation. – The hydrogen yield by dark fermentation and mixed cultures is relatively low.

8.2.6. BLACK SOLDIER FLY (BSF)

Black soldier fly (BSF) treatment is a treatment technology that involves the transformation of biowaste into insect protein and insect oil. The black soldier fly, *Hermetia illucens* L. is the fly (order: Diptera, family: Stratiomyidae). It originates from the tropical, subtropical and warm temperate zones of America. The approximate 60-70% rise in global meat consumption is expected by 2050, which can strongly influence the bioprocessing technology (Makkar et al., 2014; Lohri et al., 2017). This increase will require extensive resources that can be difficult to provide due to climatic changes, the limited availability of natural feedstock, as well as so called "food vs. fuel debate". The availability of the conventional feed (e.g. soymeal, fishmeal) can be strongly limited in the future, and their costs may be relatively high. The use of insects (including BSF larvae) as a feed can be a good solution for a quest for novel feed resources, among others, due to their high protein and lipid content. Biowaste (e.g. rotting fruit and vegetables, fish waste, animal manure, human excreta etc.) can be a feed for insects and, due to larvae life processes, can be transformed into food and feed resource (Makkar et al., 2014). The investigations conducted by Kawasaki et al. (2019) show that BSF larvae and pre-pupae can be replacement of soybean meal and oil in the diets of poultry (e.g. the laying hens). They found that the fat and chitin present in BSF may increase the eggshell thickness and values of microbiota diversity. Therefore, BSF treatment can be a sustainable way to recycle household organic wastes (Kawasaki et al., 2019). Table 8.6 presents the advantages and limitations of black soldier fly technology.

Table 8.6

Advantages and limitations of black soldier fly technology (Da Silva et al., 2020)

Advantages
– Use of BSF larvae as the feedstock generates the advantages like reduced land use, water consumption and lower greenhouse gases emissions. – Larvae can be used during production of compost for agriculture. – Larvae can be used for oil production e.g. for biodiesel. – The waste reduction with use of BSF technology is relatively high. – BSF technology has potential to reduce the viruses and pharmaceuticals in the waste. – BSF technology minimizes the environmental pollution and eliminates the odour problems.
Limitations and challenges
– Current market for BSF is unknown. – There is a possibility of heavy metal contamination of BSF prepupae for animal feeding. – The challenge in the BSF technology is the waste segregation at source. – There is a necessity for monitoring of environmental conditions and parameters of the process. – The disadvantage can be high initial investment.

8.3. THE APPLICATIONS OF THE MAJOR BIOPROCESSES

Table 8.7 presents the comparative analysis of the possible applications of the described bioprocesses which is prepared on the base of literature review.

Table 8.7

The comparative analysis of applications of main bioprocesses

Bioprocess / Application	Anaerobic digestion	Composting	Vermi-composting	Alcohol fermentation	Dark fermentation	BSF technology
Biowaste management	yes	yes	yes	yes	yes	yes
GHG reduction	yes	yes	yes	yes	yes	yes
Agriculture	yes	yes	yes	yes	yes	yes
Fertilizer and soil conditioner	yes	yes	yes	yes	yes	yes
Methane (biogas) production	yes	–	–	–	–	–
Bioethanol production	–	–	–	yes	–	–
Biohydrogen production	–	–	–	–	yes	–
Fuel	yes	–	–	yes	yes	yes
Electricity production	yes	–	–	yes	yes	–
Heat production	yes	yes	yes	yes	yes	–
Feed	–	–	–	–	–	yes

8.4. FACTORS THAT MAY INFLUENCE BIOPROCESSING TECHNOLOGY IN THE FUTURE

Based on the literature review, the major factors that may influence the bioprocessing technology in the future can be identified. These factors are dependent on climatic, economic, social, and environmental conditions. The selected factors that can influence the biological biowaste treatment technologies are described below.

8.4.1. THE NEED TO REDUCE GREENHOUSE GAS (GHG) EMISSIONS AND TO IMPROVE THE AIR QUALITY

The poor air quality in urban areas causes many premature deaths because of asthma, cardiovascular disease, lung cancer and other diseases. Therefore, better air quality and, above all, the reduction of greenhouse gas emissions is one of the reasons for countries to work together. The GHG emissions and increase of their concentration in the atmosphere has an impact on climate change, especially leading to the temperature increase. Climate change, especially global warming, has an unfavourable impact on agriculture and may lead to meteorological drought and, as a result, to agricultural drought. But on the other hand, it may result in a rise of the sea level and, as a consequence, in flooding that causes coastal problems. A very important global challenge in the future is the reduction of greenhouse gas emissions to the level that will allow stabilization of greenhouse gas concentration in the atmosphere (Erickson, 2017).

The key greenhouse gases emitted by human activities are: carbon dioxide (CO_2), methane (NH_4), nitrous oxide (N_2O), and fluorinated gases. The main economic activities that cause the production of greenhouse gases are: electricity and heat production (especially the burning of coal, oil and natural gas), industry (generally burning of fossil fuels for energy at facilities), agriculture (cultivation of crops and livestock), forestry (deforestation), transportation (mostly the burning of fossil fuels), and buildings (the burning of fuels for heat and cooking) (IPCC, 2014).

The need to reduce greenhouse gas (GHG) emissions can strongly influence bioprocessing technology in the future. Daniel-Gromke et al. (2015) found that the GHG emissions can be reduced with the use of adequate mitigation measures. They have concluded that the protection against unfavourable climate changes needs both the biowaste treatment and the quantitative assessment of the emissions related to generation of energy from biowaste. Daniel-Gromke et al. (2015) have suggested that the reduction of GHG emissions can be possible due to the avoidance of an open storage of insufficient fermented residues. GHG emissions can be also minimised due to the use of aerated post-composting with the short turnover periods, smaller heaps, and an optimized amount of the structure material.

8.4.2. INCREASE IN ENERGY DEMAND

Population growth, industrialization and urbanization are the main factors leading to an increase in energy demand. Currently, fossil fuels ensure the security of energy supply in many countries around the world, but interest in alternative fuels is gradually increasing (Rathore et al., 2019). The problems that arise in fossil-fuel dependent systems are greenhouse gas emissions, air pollution, as well as limited fuel resources (Moreno Cárdenas et al., 2019).

Reduction of fossil fuel resources and their global warming potential due to the GHG emissions calls into question their sustainability. Therefore, in the near future, more and more popular may become technologies that enable both: the obtaining of the alternative fuel as well as the decrease of greenhouse gas emissions. The example of the alternative fuel, that emerges as a sustainable fuel can be biohydrogen, produced in the process of dark fermentation. The use of biohydrogen is relatively safe for the environment because it does not liberate GHGs into the atmosphere during combustion. Additionally, an important benefit of the use of biohydrogen is its relatively low cost of production. Rathore et al. (2019) believe that the biohydrogen energy can be considered as a sustainable source of the future clean energy, taking into account the social, economic as well as environmental aspects (Rathore et al., 2019; Sekoai and Daramola, 2017).

8.4.3. GLOBAL HUNGER AND FOOD VERSUS FUEL DEBATE

Global climate change includes rising average temperatures, more severe droughts and more extreme weather variability. Global warming presents increasing challenges in agricultural ecosystems (Cheeseman, 2016). The rise of air temperatures, climate changes, and natural disasters may cause massive crop failures and can lead to food insecurity which on the other hand, can cause conflicts (including armed conflicts) and human migrations. Due to globalisation, crop failures in one part of the planet may have consequences for food security, peace, and stability in another part of the world (Mendelson Forman, 2020). An additional problem influencing food security is the continued population growth. At the same time, however, food loss, and as a consequence food waste, are huge social, economic and environmental problems. It cannot be excluded that rising food insecurity and global hunger may in the future result in the decrease of food waste (biowaste) which may have an impact on the bioprocessing technologies and, especially, on the production of biofuels (Fritsch et al., 2017; Parfitt et al., 2010).

The biofuels produced from crops or from biowaste are e.g. bioethanol (obtained in the process of ethanol fermentation) and biodiesel. They are easy applicable, renewable and biodegradable (Coma et al., 2017).The so-called first generation (1G) biofuels, are produced from edible feedstocks: grains, seeds, and sugar. However, there is a moral objection to using food-grade crops to produce fuels instead of food which leads to the so-called food versus fuel debate (Dugmore et al., 2017). Unfortunately, the large production of biofuels from crops may result in food insecurity and an increase in food prices. Additionally, food demand can lead to deforestation, ecosystem destruction, environmental impact, which can cause the loss of biodiversity. Generally, biofuel production can put stress on water resources and additionally has poor GHG benefit (Rosillo-Calle, 2012).

Therefore, a positive solution that may influence bioprocessing technology in the future is an increase in the production of second generation (2G) biofuels from substrates composed of non-edible lignocellulosic biomass whose source can be primarily agricultural and domestic organic waste. However, in the future, third generation (3G) biofuels may become more and more popular in particular. Such biofuels may be derived from the past agricultural substrates, waste vegetable oils, as well as microbes or microalgae as the alternative energy resource (Coma et al., 2017). Additionally, the prevention of the agricultural food loss should be promoted, e.g. in the form of various programs realised by countries (Fritsch et al., 2017).

8.4.4. PROGRESS IN THE DEVELOPMENT OF ENGINES FOR BIOGAS COMBUSTION

The factors that may influence the biogas economy today as well as in the future are: the process efficacy, the waste availability and logistics, as well as the properties of the end-product. AD is the technology which can have a firm commercial availability due to the access for wide variety of low-cost lignocellulosic waste. Achinas et al. (2017) in their article notice that the registration of cellulosic gaseous fuels for sale and use as a renewable fuel is the important issue in a green economy. They found that microbial activity during AD is a crucial parameter for stability of this bioprocess and the biogas yield. There is a need to modify the engines for biogas combustion because they are not yet sufficiently developed. The production of biogas will grow in the European energy market and in a few decades, it will be an economical alternative for the bioenergy production (Achinas et al., 2017).

8.4.5. POSSIBILITY OF COMBINATION OF DIFFERENT BIOPROCESSES

One of the factors that can have an impact on the bioprocessing technology in the future could be the possibility of a combination of different bioprocesses. The results of investigations conducted by Al-Rumaihi et al., (2020) show that a positive effect can be achieved through the combination of AD with composting. Biogas (methane and CO_2) and biodigestate are generated in these processes. A combination of AD and the composting may enhance their efficiency, make it possible to obtain more useful products, and have environmental benefits, e.g. generation of less waste. In this combined method, the environmental impact is compensated through the power generated by using biogas as fuel and by replacement of synthetic fertilizer by compost (Al-Rumaihi et al., 2020).

8.4.6. THE METHOD AND EFFICIENCY OF BIOWASTE COLLECTION

A very important factor affecting the bioprocessing technology in the future can be the method and efficiency of biowaste collection, especially the share of biowaste that will be collected selectively. Pavlas et al. (2020) suggest that a sustainable way of handling biowaste is their separate collection and treatment by composting or fermentation. They emphasize that the biowaste treatment may have the environmental benefits and must be further explored (Pavlas et al., 2020). Sorting efficiency is the amount of correctly sorted out biowaste expressed as the percentage of the total yield of biowaste. This parameter is influenced by collection equipment as well as social and economic factors. One of these factors can be e.g. the degree of awareness about the significance of sustainable biowaste handling, living-conditions, as well as economic conditions (Hansen, 1996). The behaviour of consumers is an important factor influencing waste processing efficiency. The selectively collected biowaste can be used e.g. as a valuable resource for biofuel production through fermentation due to its organic and nutrient rich composition (Wojnowska-Baryła and Bernat, 2020).

8.4.7. THE NEED OF INVESTIGATION FOR NEW RENEWABLE MATERIALS

Environmental problems that originated from petroleum materials and products have led to the investigation for new renewable materials (e.g biopolymers) that can be produced from biomass (e.g. biowaste). These materials (biopolymers, bioplastics) may be used as compostable packaging and could become be more and more popular in the future. The example of totally renewable, biodegradable, and biocompatible aliphatic polyesters are polyhydroxyalkanoates (PHAs) which are synthesized from glucose-rich substrates in the cytoplasm of some bacteria, and polylactic acid (PLA) synthesized from bio-derived monomers (Figueroa-Lopez et al., 2019; Wojnowska-Baryła and Bernat, 2020). Some PHAs can be used for production of bioplastic packaging articles, among others food packaging and service ware. Such bioplastic active packages can improve food conditions and extend its shelf life. Packaging articles made from PHA can interact with food and the internal packaging environment. They would release or absorb the specified components into or from food. These active properties can be achieved due to the addition of substances having the antioxidant and antimicrobial properties, for example: essential oils and natural extracts derived from biowaste (Figueroa-Lopez et al., 2019).

8.5. CONCLUSIONS

These above-described factors depend on climatic, economic as well as social and environmental conditions. Additionally, biowaste treatment technologies may be strongly affected by pollutants and xenobiotics present in the biowaste (e.g. heavy metals, pharmaceuticals, microplastic etc.) which can complicate its processing and valorization. A very important factor affecting bioprocessing technology in the future seems to be climate change, especially global warming, resulting i.e. from the GHG emissions, deforestation and urbanization. Global warming can lead to natural disasters, global drought, but at the same time it contributes to heavy storms, floods, hurricanes etc. As a result, global warming leads to the degradation of water resources and crops whose consequence can be food scarcity in many places around the world. This, on the other hand, may be a reason of poverty, local conflicts, and human migration. However, it should be emphasized that there are many factors that are not always foreseeable (e.g. the COVID-19 pandemic and the treatment of other epidemics that can be possible in the future) whose impact on economics, politics, human health, the environment and, as a result, on the quantity and quality of biowaste and bioprocessing technologies, can only be known in the near or distant future.

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Part III

Biowaste/sewage sludge: occurrence and environmental risk in the case of application on soil

Chapter 9

Evaluation of the effectiveness of currently used methods of identification of microorganisms in soils

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9.1. INTRODUCTION

Microorganisms have numerous characteristics that allow them to develop in all habitable environments. One of the areas with high biological activity is the soil environment. The microorganisms present in this environment influence the process of shaping soil fertility, make nutrients available, and take part in the removal of harmful chemicals. Therefore, they are one of the main soil-forming factors.

The soil microbiome has been the subject of numerous studies over many years but is still not fully understood. The microorganisms included in the microbiome are highly diverse, both structurally and functionally. They play a major role in the biogeochemical processes occurring in the soil. Furthermore, most of the soil microorganisms are incapable of being grown and maintained under artificial conditions. This makes it difficult to identify them. It is known that the soil is one of the richest habitats for microorganisms. A fertile soil environment can contain even billions of bacteria per 1 g of fresh matter, including thousands of different species (Nannipieri et al., 2020).

It is estimated that only about 5% of the fungi species and about 12% of the bacteria species found in the environment are currently known. The diversity of microorganisms is determined by genetic, species, and functional diversity. A greater diversity of microorganisms occurs when the environment is richer in nutrients and chemical substances. Knowledge of the structural and functional diversity of microorganisms makes it possible to determine the condition of the soil environment, which is extremely important for agriculture and ecology (Kozdrój, 2013).

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9.2. METHODS USED TO IDENTIFY MICROORGANISMS IN THE SOIL

The methods used in microbiological laboratories can be divided into conventional methods, based on microscopy techniques, cultures on microbiological and biochemical plates, and methods based on molecular techniques.

The conventional methods include: microscopic examination, culture, and identification of microorganisms based on biochemical characteristics on growth media. It is estimated that only about 1% of the soil microbial population can be isolated using conventional laboratory methods. For this reason, modern techniques are increasingly being used to study bacterial and fungal populations in this environment, with particular emphasis on molecular biology. Molecular methods use different techniques to detect characteristic DNA or RNA sequences of microorganisms. Several methods can be used for the identification of microorganisms and microbiological activity in soils, classified according to the type of techniques used or the purpose of analyses (Gałązka et al., 2016; Torsvik and Ovreas, 2002).

Table 9.1 illustrates the methods used to determine the structural and functional diversity of soil microorganisms (Furtak and Gajda, 2018; Gałązka et al., 2016; Kozdrój, 2013).

Table 9.1

Methods of determination of the structural and functional diversity of a soil microbiome

<table>
<tr><th colspan="2">Structural analysis</th><th colspan="2">Functional analysis</th></tr>
<tr><td rowspan="2">Analysis of fatty acid profiles</td><td>PLFA</td><td rowspan="3">Analysis of carbon source utilization</td><td rowspan="2">standard microbiological media</td></tr>
<tr><td>FAME</td></tr>
<tr><td rowspan="3">Nucleic acid analysis</td><td>DNA sequencing</td><td>ECOplate ™</td></tr>
<tr><td>methods based on PCR</td><td rowspan="4">Enzyme activity measurements</td><td>Spectrophotometric</td></tr>
<tr><td>hybridization techniques</td><td>Fluorimetric</td></tr>
<tr><td rowspan="3">Protein profile analysis</td><td>liquid chromatography</td><td>Calorimetric</td></tr>
<tr><td>two-dimensional electrophoresis</td><td>Chromatographic</td></tr>
<tr><td>isotopic and isobaric determination</td><td colspan="2">Chromatographic measurements of the volume of gas released (e.g. methane, hydrogen dioxide)</td></tr>
<tr><td rowspan="2">Determination of microbial counts</td><td>plate cultures</td><td rowspan="2">Determination of microbial counts</td><td>plate cultures</td></tr>
<tr><td>based on the qPCR technique</td><td>based on the qPCR technique</td></tr>
</table>

9.3. PURPOSE AND SCOPE OF THE STUDY

Several methods can be used in the study of microbiological diversity, both conventional and molecular. Methods of identification of soil microorganisms are constantly being improved by researchers in many countries. Modifications are also being developed and new methods are being introduced.

This study discusses research methods that allow for the analysis of microbial differentiation in the soil environment. Conventional (microscopic), biochemical, and molecular methods used for structural analysis of the microbiome were presented.

9.4. CONVENTIONAL METHODS OF IDENTIFICATION OF MICROORGANISMS

The conventional microbiological analysis allows for the determination of the number of microorganisms and their identification based on plate cultures. To date, these are the most common methods used in soil microbiology to identify microorganisms and are based on time-consuming and labor-intensive techniques of passage and culture in incubators at a specific temperature. The next stage is to count colonies of microorganisms growing on non-selective and selective microbiological media.

Culturing is conducted under conditions most suitable for the growth of the microorganisms analyzed. Microorganisms can grow under very different conditions of temperature, pH, humidity, osmotic pressure, and oxygen content. The serial dilution method is often used. This method allows for evaluation of the general count of e.g. bacteria (vegetative and spore forms), actinobacteria, and fungi. The reading of the quantitative analysis is performed after an incubation period specific to each group of microorganisms. The number of colony-forming units (CFU) is determined, which corresponds to the number of microorganisms in a given volume of the soil analyzed (Fig. 91a). After quantitative analysis, in order to assign organisms to specific species, isolation is conducted on specific agars, most often selective. After the incubation period and under appropriate conditions, vital microscopic preparations are prepared from the cultures obtained and observed under a light microscope. The microscopic examination allows for the observation of the morphology of microbial cells (Fig. 91b). Microorganisms are most often identified based on the appearance of a single colony using the keys to determine systematic affiliation (Galus-Barchan and Paśmionka, 2014).

Furthermore, conventional biochemical techniques of microbial analysis consist in the observation of changes occurring during chemical reactions. The results include a change in the color of the medium, gas production, or a change in the color of the solution being tested.

Biochemical methods allow for the analysis of the activity and taxonomic identification of microorganisms, also based on the determination of the products of their metabolism or selected compounds that form part of cell structures. One limitation in using conventional methods is their low accuracy. It is estimated that they allow for isolation of only ca. 1% of the microorganisms that are present in the soil. This group of microorganisms belongs to the organisms requiring complex nutrients (Furtak and Gajda, 2018).

a) b)

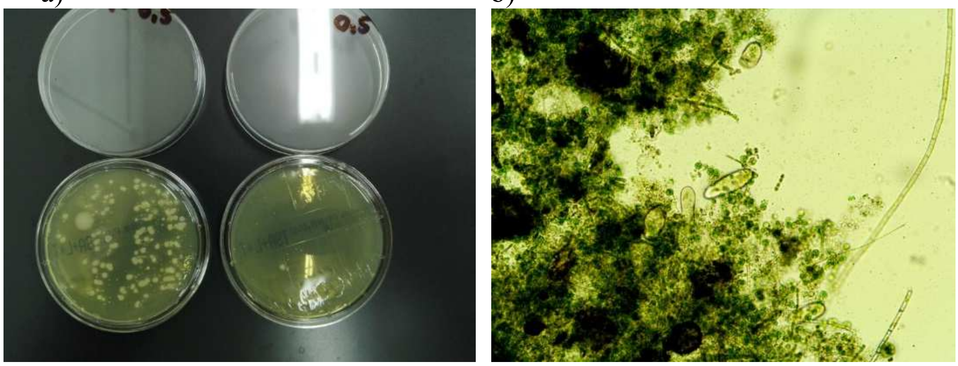

Fig. 9.1. Identification of microorganisms: a) the Koch plate method, b) microscopic method (author's photos)

9.5. MODERN TECHNIQUES FOR IDENTIFICATION OF MICROORGANISMS

9.5.1. ANALYSIS OF FATTY ACID PROFILES

Fatty acid analysis is a biochemical-physical method that determines the activity and structure of microorganisms, without the need to culture them. Phospholipids are present in every living cell in cell membranes but are not present in spare products or dead cells. The determination of fatty acids, which are part of cell structures, allows for the taxonomic identification of microorganisms. These acids represent characteristic biomarkers due to their high structural diversity and biological specificity. Typically, certain types of fatty acids dominate in individual taxa. Fatty acids are very useful in the systematics of microorganisms and are used in the creation of databases for the identification of microorganisms.

The method of analysis using fatty acids consists in their isolation from the soil, extraction in organic solvents, followed by the analysis using gas chromatography. Two main methods are used:

- ➢ phospholipid fatty acid analysis (PLFA),
- ➢ fatty acid methyl ester analysis (FAME).

When comparing the two methods, it should be stressed that the former is more time-consuming and requires more research material. However, the advantage of this analysis is that only fatty acids from living cells are taken into consideration. Furthermore, the FAME method requires less material and less time (one day, whereas 6 days are needed to perform PLFA analysis). However, the analysis determines all the lipids present in the soil, including those from dead animal and plant biomass. Therefore, it cannot be used for reliable structural analysis of living and biologically active soil microorganisms (Dunfield et al., 2008; Furtak and Gajda, 2018; Smithwick et al., 2005).

9.5.2. NUCLEIC ACID ANALYSIS

The DNA sequencing method is currently an irreplaceable tool for creating genetic libraries covering many plants and animals, including microorganisms. The analysis of soil microbial biodiversity is a rapidly growing field of genetic research. It provides more and more insights into the microbiological composition of the soil environment. Using modern molecular biology techniques does not require culturing microorganisms on special media. This is all the more important because some researchers estimate that up to 99% of microorganisms are not cultured using modern methods. Like any analytical technique, sequencing also has some drawbacks and imperfections, but this method has contributed to a significant development in science and has allowed for the analysis of entirely new elements, including those which are not examined (Hodkinson and Grice, 2015).

An important advantage of sequencing was the use of next-generation sequencing (NGS) due to process automation and commercialization. The process is conducted in special sequencing devices. Continuous development of the method has made it possible to produce efficient and user-friendly devices at a relatively low price. Next-generation sequencing methods used in modern laboratories include:

- pyrosequencing 454 (454 Life Sciences),
- Solex sequencing (Illumina),
- Solid method (Applied Biosystems company) (Gałązka et al., 2016).

The first method slowly ceases to be used due to its limitations. Since DNA is divided into very small fragments, problems arise when reading the whole genome. The method can be particularly unreliable for unknown species (Hodkinson and Grice, 2015).

The second technology combines Sanger sequencing with pyrosequencing. It is commonly used for organisms living in the soil environment. The method is currently developing very rapidly and is constantly being improved. It enables fast sequencing, even without having reference sequences (Illumnia, 2016; 2017).

The third technique is characterized by a very high reading accuracy of 99.9%, due to the use of a system of probes, labeled with fluorescent markers (Liu et al., 2012).

Metagenomics, which is a study that involves the sequencing and construction of genome libraries, allows for the identification of soil bacteria based on the analysis of the 16S rRNA sequence. It is of particular importance in the case of microorganisms that cannot be cultured and allows for the tracking of changes occurring in the natural environment.

In the analysis of nucleic acids, gene replication techniques are mainly used, based on the polymerase chain reaction (PCR) technique. Although the PCR technique was already developed in the 1980s, it wasn't until the late 1990s that it gained popularity. It consists of multiple duplications of any DNA sequence using so-called primers. These are short DNA oligonucleotides (about 20 nucleotides in length), complementary to the array fragments at both ends of the gene. There are two types of primers: a forward primer, whose sequence must be the same as the duplicated sequence, and a backward primer, whose sequence must be complementary to the duplicated sequence. The reaction mixture consists of the following components: array DNA (duplicated DNA isolated from the test sample), excess nucleotides (deoxyribonucleoside triphosphates), excess primers, and thermostable DNA polymerase.

This composition of the mixture ensures the correct course of the reaction. The PCR reaction itself consists in repeating the cycle of three stages several times (Fig. 9.2):

- thermal denaturation of the tested DNA at a temperature of about 95°C,
- connecting primers to the array at a temperature reduced to 40-60°C (hybridization),
- polymerization, i.e. synthesis of new DNA strands at 72°C using special thermostable polymerase (elongation).

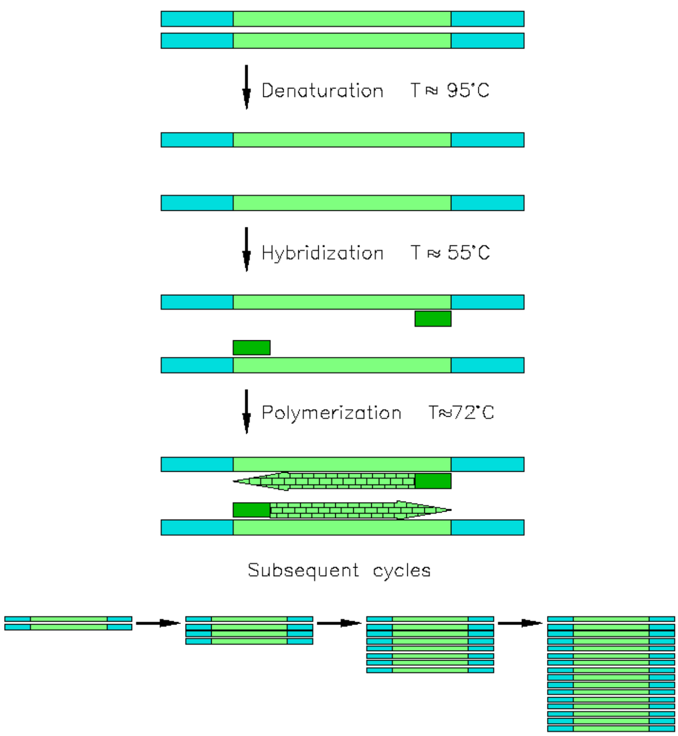

Fig. 9.2. PCR reaction scheme

The number of cycles ranges from about 30 to 40. Each PCR reaction shall be conducted simultaneously for test samples and controls, both positive and negative. In the last phase, a sample of the reaction mixture is subjected to an electrophoresis process to visualize DNA molecules of adequate size, corresponding to the size of the DNA fragment between the primers (Kondak, 2009; Nowakowska, 2006).

A more recent version of the PCR method is the real-time polymerase chain reaction (PCR). The reaction is carried out in special devices, using specially prepared primers, also called probes. Reading the results of the PCR reaction is possible in subsequent cycles and allows for the determination of the amount of array used during the reaction. To this end, the level of sample fluorescence is measured, which is proportional to the amount of product.

Both varieties of the method, i.e. both conventional PCR and real-time PCR, are currently used to determine the presence of microorganisms (bacteria, viruses, fungi) in the test sample.

Reagent kits of different manufacturers are available on the market, making it possible to conduct conventional PCR or real-time PCR reactions after isolation of DNA from the sample. However, such determination requires appropriate laboratory equipment and competent and trained personnel.

The most commonly used molecular marker methods, based on polymerase chain reaction (PCR), include: randomly amplified polymorphic DNA (RAPD), sequence characterized amplified regions (SCAR), amplified fragment length polymorphism (AFLP), and simple sequence repeats (SSR).

The RAPD method is based on the use of a random (usually 10-nucleotide) primer for amplification. After the reaction and electrophoretic analysis, different band profiles are obtained, which are a characteristic feature of a given strain of microorganisms. A comparison of the profiles allows for the determination of the level of similarity between the strains tested. The advantage of the method is its speed, universality, and the fact that it does not require knowledge of the DNA sequence of the examined microorganism. The method is also relatively inexpensive. Unfortunately, due to the low temperature of connecting primers (necessary to get a sufficient number of products), the technique has low repeatability and is more sensitive to changes in reaction conditions than normal PCR. The results are affected by many factors that are difficult to determine (for example, small changes in temperature, heating or cooling time, reagents used). However, it can be successfully used for preliminary research on molecular differentiation of a specific species, e.g. using the SCAR technique (Nowakowska, 2006; Rastogi and Sani, 2011; Sztuba-Solińska, 2005).

The AFLP method combines the advantages of PCR and RFLP (described below). The first stage consists in cleaving the purified DNA with restriction enzymes. This technique uses two enzymes: one with higher and one with lower digestion frequency. The obtained restriction fragments are accompanied by adaptors, consisting of a core sequence and a sequence specific to the restriction site. The second stage is characterized by the proper selective amplification of fragments. It is conducted using primers that differ from those of the first phase with the presence of two additional selective nucleotides. Next, the separation is carried out on polyacrylamide gel. After an exposure time of about 24 hours, an autoradiograph is obtained, generally consisting of about 50 to 100 bands. The analysis of the results is carried out using computer software due to the high density of bands. The advantages of the technique include the ability to identify a large number of markers in a relatively short time, high repeatability, very good resolution of the obtained band patterns, and the possibility of automation. At the same time, the required amount of initial material is relatively small. The disadvantage of the method is the high cost (Sztuba-Solińska, 2005).

The SSR technique uses amplified microsatellite sequences. Microsatellites are found on all chromosomes, in both coding and non-coding regions of the genome, in prokaryotic and eukaryotic organisms. They are strongly polymorphic due to the different lengths and numbers of repetitions of the basic sequence. Microsatellites are detected with PCR reaction, using two primers flanking a repeatable sequence. The products resulting from amplification are separated in the acrylamide gel using electrophoresis. The results are analyzed using bioinformatics methods. Due to the high genomic specificity, markers are developed individually for each species. They are successfully used for species identification, gene mapping, and examination of the population structure and phylogeny. The disadvantage of the method is the necessity to know the flanking sequences in the case of SSR polymorphism. The cost of analysis is also high and the process is difficult to automate (Nowakowska, 2006; Scott et al., 2012; Sztuba-Solińska, 2005).

Another technique based on the PCR method is polymerase chain reaction – restriction fragments length polymorphism (PCR-RFLP). In this method, a selected fragment of DNA, specific to the microorganism, is amplified by PCR, and then restriction enzymes are used to identify the DNA mutation. These are bacterial enzymes, capable of cleaving the double strand of DNA at specific locations. In the case of even a small change in the recognized sequence, the strand digestion process will not take place. Therefore, restriction enzymes have become a very important and frequently used tool in molecular techniques (Fakruddin et al., 2013; Gałązka et al., 2016; Marciniak and Robak, 2012).

Terminal restriction fragment length polymorphism (T-RFLP) represents a modification and extension of the PCR-RFLP method. It allows for the analysis of microbiological diversity in less time. In this method, one of the primers is labeled with a fluorescent dye.

The advantage of both techniques (PCR-RFLP and T-RFLP) are high reliability and reproducibility of results. The problem with their use is the proper choice of primers for the polymerase reaction and the need to obtain a large amount of isolated high-quality DNA. These techniques are also time-consuming and expensive. However, the analysis of a set of genes allows for the determination of the species structure and provides knowledge on phylogenetic relations of the examined microorganism complex. This makes it possible to assess the biodiversity of the soil analyzed (Gałązka et al., 2016; Kumar and Josih, 2015; Liu et al., 1997; Sztuba-Solińska, 2005).

Similar capabilities are offered by the techniques based on ribosome analysis. These include: ribosomal intergenic spacer analysis (RISA), automated ribosomal intergenic spacer analysis (ARISA) and amplified ribosomal DNA restriction analysis (ARDRA) used to examine environmental samples. The first technique is quite often used to determine the dominant microorganisms in the material. However, large amounts of DNA are needed to obtain reliable results. It is also time-consuming and the silver staining method is characterized by low sensitivity (Gałązka et al., 2016; Kirk et al., 2004; Sztuba-Solińska, 2005). For this reason, a modification was added in the form of the ARISA technique, in which the analysis is performed automatically using a laser detector and fluorescent-labeled primers. This improved the sensitivity of the method and reduced the time needed for the analysis. The disadvantages result from the limitations of the PCR technique itself and possible background fluorescence (Liu et al., 1997; Fisher and Triplett, 1999; Fakruddin et al., 2013). The third technique (ARDRA) is also often used to evaluate the diversity of microorganisms. This method uses additional cloning of the amplified gene. Analyses have shown its higher efficacy compared to randomly amplified polymorphic DNA (RAPD) or amplified fragment length polymorphism (AFLP) techniques (Dherbecourt et al., 2006; Koeleman et al., 1998; Pandey et al., 2009).

It seems that DNA microarrays will also be used in the future to detect microorganisms. A microarray is a glass plate with DNA fragments deposited using a regular pattern, with different sequences. These fragments represent probes, which, through hybridization, detect complementary DNA or RNA molecules. However, microarrays are currently mainly used to examine gene expression (Żabicka and Literacka, 2013).

9.5.3. PROTEIN PROFILE ANALYSIS

This method belongs to the molecular methods and is based on the determination of the quantitative and qualitative composition of proteins, allowing for the determination of the species affiliation of the microorganisms studied. The result is compared with a database that collects data on the spatial structure of proteins and nucleic acids for already identified organisms. The largest and most popular base is the Protein Data Bank, which, as of 1 May 2020, contained 163,633 protein structures.

The following techniques are used in protein profile analysis:

- two-dimensional gel electrophoresis,
- liquid chromatography with mass spectrometry,
- isotope-coded affinity tag (ICAT) and isobaric tag for relative and absolute quantitation (iTRAQ).

Two-dimensional gel electrophoresis is the first developed method of protein profile analysis. The most commonly used SDS-PAGE analysis includes isolation of proteins, electrophoretic separation in polyacrylamide gel, and staining of obtained proteins. The use of fluorescent dyes improved the sensitivity of the method. However, a disadvantage of the analysis is that a maximum of several hundred proteins can be recognized. Therefore, only proteins of low complexity are identifiable using this method.

Liquid chromatography does not require the use of gel. However, much like the gel electrophoresis method, it allows for the determination of a maximum of 20 to 40% of the proteome.

Therefore, both methods allow for the identification of microorganisms but do not provide quantitative data. Such data can be obtained by isotope-coded affinity tagging (ICAT) and isobaric tag for relative and absolute quantitation (iTRAQ) (Furtak and Gajda, 2018; Zhang et al., 2010).

The limitation of all the above methods is the difficulty related to the subject of the research. The protein structure is spatially complicated. An additional challenge is their presence in multimolecular complexes. Changes in protein profiles also occur under the influence of environmental factors, e.g. changes in soil pH, the abundance of nutrients, and temperature.

Nevertheless, the analysis of protein profiles is a dynamically developing method. The aim is now to identify specific protein biomarkers that would allow for the identification of microorganisms without the need to analyze the full structure.

Mass spectrometry (e.g. MALDI-TOF MS) has recently become increasingly common in microbiological diagnostics. It is currently used to identify selected strains in environmental samples. The method is based on the analysis of proteins occurring in large quantities, such as ribosomal proteins, and allows for both the identification of microorganisms (bacteria, yeast-like fungi, filamentous fungi) and the analysis of affinity within isolates of the same species. These proteins are found in every microorganism and represent its genetic fingerprint. The protein spectra are compared with a constantly updated library and, based on this information, the microorganism is identified. The advantage of the technique is the short time of analysis, ranging from a few minutes for a single sample to about 1.5 hours for a 96-sample plate. Material containing a small amount of initial material (a single colony or liquid culture sample) is sufficient for identification. The disadvantage of the method is its high cost and the need to purchase specialized equipment (Żabicka and Literacka, 2013).

9.6. SUMMARY AND CONCLUDING REMARKS

Any change in soil properties has an effect on the number and activity of the microbiome. The large genetic diversity of soil microorganisms may be the source of new strains, which will allow obtaining new, natural products and will contribute to the modification of the already existing or new industrial technologies. However, it is necessary to use appropriate methods for the identification of microorganisms.

The methods outlined above may complement each other. It should be remembered that in the case of soil, it is not only the presence of microorganisms (determined based on quantitative and qualitative analysis) that is important but also the determination of their functions. Therefore, structural analysis is usually combined with functional analysis. Only with such correlation of the analyses, including both identification of microorganisms and determination of their activity, can the quality and functionality of the soil environment be determined.

Conventional research techniques do not allow for a full analysis of the variability of the microbiological composition of soils. For this reason, molecular methods based on e.g. PCR reaction are increasingly used for this purpose. They represent a real breakthrough in the applied research methods and enable the discovery of new previously unidentified microorganisms. This is particularly important in the case of viable but non-culturable (VBNC) microorganisms. Modern research techniques, based on the genetic fingerprinting and PCR reaction methods, are among the best tools used for the identification, detection, and quantitative determination of microorganisms. They are widely used in research on biodiversity and genetic variability. Molecular methods are very precise, sensitive, and much more specific than conventional methods. Analyses using molecular biology are fast, and the results are reproducible and reliable.

However, it should be considered that according to the opinion of many microbiologists, despite the use of advanced analytical methods, up to a maximum of 10% of the microorganisms present in the soil can be isolated.

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Chapter 10

Determining the relationship between the properties of sewage sludge/biowaste introduced into soils and the level of microbiological activity of soils

Beata BIEŃ

10.1. INTRODUCTION

Biowaste, according to the Waste Framework Directive (2008/98/EC), means biodegradable garden and park waste, food and kitchen waste from households, offices, restaurants, wholesale, canteens, caterers and retail premises and comparable waste from food processing plants. This definition does not include waste from forestry and agriculture, manure, sewage sludge or other biodegradable waste (eg. natural textiles, paper or processed wood). Biodegradable waste is a broader term that, as defined in the Landfill Directive (1999/31/EC), covers all wastes that may undergo aerobic or anaerobic digestion processes, such as food and garden waste, paper, and cardboard. In the EU, biowaste usually accounts for 30 to 40% of municipal solid waste – but this range can expand from 18% to as much as 60% (Jędrczak, 2018).

Biowaste and biodegradable waste can be processed by various forms of treatment (biological), however composting and digestion are among the most commonly used processes. Municipal sewage sludge shows a large variability in chemical composition, which depends on the properties of the waste, its treatment technology and processing. Among the various methods of processing sewage sludge into fertilizer, composting together with the municipal biowaste fraction, straw, bark, chips, garden waste or food industry is mainly used because of the simplest and cheapest method (Górska and Stępień, 2008; Krzywy et al., 2008).

Municipal sewage sludge is usually characterized by high soil-forming and fertilizing values. The natural use of sewage sludge, in addition to the beneficial effect on the amount of micro- and macroelements in soil (source of nitrogen, phosphorus, potassium, calcium and magnesium), improves soil physical properties, increasing its sorption capacity due to the presence of organic matter in sewage sludge (Gawdzik, 2010). In addition, sewage sludge causes changes in the

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structure of soil microorganisms. Organic and mineral compounds introduced into the soil together with sewage sludge have a significant impact on the number of microorganisms and changes with the participation of enzymes whose activity changes in soil (Nowak et al., 2010; Sullivan et al., 2005). The addition of sewage sludge with low heavy metal contents positively affects both the biomass increase and the number of soil microorganisms (Fijałkowski et al., 2012; Grobelak et al., 2013; Nannipieri et al., 2003, Placek et al., 2014;). Therefore, determining changes occurring in the number of microorganisms, soil enzymatic activity and the intensity of processes related to carbon and nitrogen circulation after adding sewage sludge, bio-waste and determining their impact on soil fertility is an important issue (Grobelak et al., 2013; Jezierska-Tys and Frąc, 2008). Restoring components accumulated in sewage sludge and biowaste to soil is not only economically appropriate but also necessary to preserve and restore ecological balance. Unfortunately, sewage sludge cannot always be applied for fertilization. The problem associated with the natural use of municipal sewage sludge is a high content of heavy metals. They can cause changes in soil fertility, reduce crop yielding and affect crop quality, and may pose a threat to the consumer (Kazanowska and Szaciło, 2012; Rosik-Dulewska et al., 2007). Therefore, the use of sewage sludge for fertilization and reclamation of biologically weak and degraded soils is reasonable (Jezierska-Tys and Frąc, 2008). Sludge for non-industrial use should meet the relevant requirements due to its toxic effects on living organisms and the possibility of bioaccumulation (Grobelak et al., 2013; Kacprzak et al., 2014). The aim of the chapter is to discuss the properties of sewage sludge and biowaste introduced into soil, its microbiological activity and the analysis of the relationship between them.

10.2. PROPERTIES OF SEWAGE SLUDGE AND BIOWASTE IN TERMS OF THEIR NATURAL APPLICATION

The amount and physicochemical and biological composition of the sludge is varied and depends on the quality of the sewage, treatment plant processes and methods of sludge treatment (Malej and Majewski, 2002). The use of waste rich in organic matter e.g. sewage sludge, straw, sawdust, improves the physicochemical conditions of soil and the balance of humus compounds (Nowak et al., 2001). Sewage sludge is a waste rich in organic matter (Czekała, 2002). Non-stabilized sludge contains from 75 to 85% and stabilized from 30 to 50% organic carbon calculated on the dry matter (Rosik-Dulewska, 2002). The impact of sewage sludge on soil humus depends not only on the content of organic matter, but also on the nitrogen content. The amount of nitrogen in municipal sewage sludge is 2.5% on average, with fluctuations from 0.9 to 7.6% on a dry weight basis (Rosik-Dulewska, 2002). The ratio of organic carbon to nitrogen (C:N) in soil is an important parameter of the assessment of soil habitat conditions, and thus of the quality of humus (Wiater and Dębicki, 1993). The carbon to nitrogen (C:N) ratio in humic acids is as 10.5-17.2:1. The value of the C:N ratio can affect the dynamics and time of sludge decomposition in soil environment, because it is one of the important parameters affecting the decomposition of organic matter and the humification of the formation of humus compounds (Mazur, 1996). In municipal sewage sludge the carbon to nitrogen ratio is as 10-13:1 (Fidecki, 2002; Filipek and Fidecki, 1999). The carbon to nitrogen ratio of sewage sludge can be considered as close to soil humus and can contribute to the full utilization of this waste in the soil environment (Krzanowski et al., 1992). The element that plays an important role in plant nutrition is phosphorus (Sapek and Sapek, 1999). The amount of phosphorus in municipal sewage sludge is in the range of 0.6-9.2% in dry matter, on average about 3%, and is similar or slightly higher than in natural organic fertilizers (Rosik--Dulewska, 2002; Siuta, 2002). The phosphorus content in sewage sludge is more stable than nitrogen because its compounds are poorly soluble. Potassium is the scarcest constituent of sludge (Czekała, 2002). This is due to the easy solubility of potassium compounds, the content of which increases in sludge waters (Skorbiłowicz, 2002). The potassium content in sewage sludge is 0.1-0.6% of dry matter and is always lower than it is in manure and other traditional organic fertilizers (Rosik-Dulewska, 2002).Therefore, during agricultural use of sludge, fertilization should be supplemented with potassium (Czekała, 2002). The calcium content in sewage sludge ranges from 1 to 10% of dry matter, and on average is 2.5%. It is over 1.5 times larger than in manure. The presence of this component in the sludge largely depends on the share and nature of industrial wastewater flowing into the wastewater treatment plant (Rosik-Dulewska, 2002; Skorbiłowicz, 2002). Municipal sewage sludge has a magnesium content of 0.5 to 1%, which is sufficient to supplement magnesium deficiency in soil, as well as to provide plants with this nutrient (Czekała, 2002; Siuta, 2002; Skorbiłowicz, 2002). Sewage sludge is also an important source of sulfur, as its content is on average 1.1% of dry matter (Czekała, 2002).

In addition to valuable components, sewage sludge may also contain many harmful substances (Baran and Oleszczuk, 2002; Babel et al., 2006; Grzywnowicz and Strutyński, 2000). These undesirable components include: heavy metals (Bozkurt and Yarilgac, 2003; Shrivastava and Banerjee, 2004), polycyclic aromatic hydrocarbons (Czekała et al., 2002; Harrison et al., 2006) and other toxic organic compounds (Harrison et al., 2006). The main elements whose occurrence has an adverse effect on living organisms are: mercury, cadmium, nickel, chromium, and lead. With the exception of nickel, these metals exhibit relatively high bioaccumulation. Copper, zinc, cobalt, molybdenum, and boron (microelements) are necessary for the proper growth of plants, but too high a concentration of these metals in the soil may negatively affect living organisms (Podedworna and Umiejewska, 2008). Some of the heavy metals cannot be completely removed from the natural cycle because they are considered microelements. It is necessary to determine the bioavailability of the heavy metal form.

The management of sewage sludge is determined not only by the physicochemical composition, but also by its hygiene and sanitation, usually determined on the basis of indicator organisms, such as bacteria from the *Coli* group, *Clostridium perfringens*, fecal streptococci and helmint eggs (Malej, 2000; Institute of Environmental Engineering, 2004). Sewage sludge is inhabited by numerous microorganisms and microfauna, creating a specific biocenosis. It consists of bacteria, viruses, parasitic worms, fungi, protozoa, and others. Among these microorganisms, there are both pathogenic, dangerous to human and saprophytic microorganisms, neutral from the sanitary point of view (Bień, 2002; Krzywy and Iżewska, 2004; Rosik-Dulewska, 2002). An additional problem in sewage sludge is the presence of pathogenic microorganisms from the genera Enterobacteriaceae (Salmonella, Shigella) and fungi which are dangerous for health (Institute of Environmental Engineering, 2004).

Many scientific papers (Budzińska et al., 2009; Loc, 2002; Kocaer et al., 2004; Joniec and Furczak, 2007) indicate that a significant proportion of microbial contaminants are pathogenic bacteria. Bacteria belonging to the species such as *Escherichia coli*, *Clostridium perfringens*, *Bacillus anthracis*, *Listeria monocytogenes*, *Vibrio cholerae*, *Mycobacterium tuberculosis*, *Streptococcus faecalis*, *Proteus vulgaris* and the genus *Salmonella* sp., *Shigella* sp. can lead to contamination of water and become a real threat to the health and life of animals and humans (Kocaer et al., 2004; Loc, 2002; Pepper et al., 2006). The presence of pathogenic fungi is also a limitation in the use of sewage sludge. An important threat are also viruses contained in sewage sludge (Jezierska-Tys and Frąc, 2008). The greatest threat is posed by viruses belonging to the enterovirus group, which are characterized by high resistance to disinfectants and have a long infectious capacity in the environment (Bień, 2002; Pepper et al., 2006). In addition to microbiological indicators, an important criterion is the presence of gastrointestinal parasites eggs (Pepper et al., 2006).

Among the gastrointestinal parasites present in sewage sludge, tapeworms, nematodes, and flukes have epidemiological significance (Bień, 2002; Jezierska--Tys and Frąc, 2008;).

Therefore, in the case of the natural use of sludge, it is important to know sludge stabilization state from a sanitary and hygienic point of view before using sludge in agricultural production or in reclamation (Budzińska et al., 2009; Joniec and Furczak, 2007).

Annually, 118-138 million tonnes of biowaste is generated in the European Union, but less than 40% is recycled (Krzanowski et al., 1992). Biowaste usually accounts for 30 to 40% of municipal solid waste - this range can, however, expand from 18% to as much as 60% (Jędrczak, 2018). The average biowaste production per capita in 2019 was around 170 kg. The smallest amount (17.5 kg/person) of biowaste was achieved in Poland, Hungary, Estonia, and Portugal. The largest amount (320 kg/person) was in Slovenia (Giacomazzi, 2019). The biological solid waste fraction consists of two main streams: waste from green park or garden and kitchen waste. The first usually contains 50-60% water, as well as more wood (lignocellulose), the second does not contain wood, but up to 80% water. The composition of the biological solid waste fraction varies between cities and countries but always contains a significant amount of biological material.

The amount of biowaste from municipal sources depends mainly on the amount of green waste. A characteristic feature of biowaste is high humidity, usually exceeding 50% (52-80%). Organic substances constitute from 34 to 81% of their dry matter, the C:N ratio is 10-25, and the biogas potential is 0.15-0.60 m^3/kg DS. It is estimated that about 30% of the biowaste potential (27 million Mg) is green waste. The term "biowaste" often refers only to kitchen waste, excluding green waste. Kitchen waste consists mainly of food waste. Other biodegradable waste that can be composted alone or together with the biodegradable fraction of solid waste are primarily the following elements:

- commercial food waste, including:
  - waste from markets,
  - catering waste;
- forestry waste, including:
  - bark,
  - leftover wood;
- agricultural waste, including:
  - animal droppings (solid and liquid manure),
  - straw waste,
  - leftovers from growing beans, peas, flax, and vegetables,
  - used mycelium compost,
- wastes from the food and drink processing industry, including:
  - distilleries and malt factories,
  - wine cellar,
  - fruit and vegetable processing industry,
  - potato processing industry, including starch production plants,
  - leftovers from sugar beet production,
  - slaughterhouse waste,
  - meat production waste,
- sewage sludge (obtained through biological treatment of municipal wastewater).

The use of compost from biowaste in soil as an organic fertilizer or soil improver has various environmental implications. When compost is used in soil, the chemical components of compost penetrate the soil. In particular, heavy metals and some organic pollutants should be considered in the context of negative impacts.

The heavy metal content in compost depends mostly on the input materials. However, what happens to heavy metals introduced into the soil with compost depends on a number of factors, such as crop characteristics and soil type and pH (Saveyn and Eder, 2013). Repeated applications of compost can lead to the accumulation of heavy metals, and the long-term effect of such accumulation has not been fully understood yet. Scientific literature reviews (Smith, 2009) show only the positive effects of compost use on the microbiological state and soil fertility. However, the accumulation of heavy metals depends on local differences (overall background concentrations generally increase). The problem is the ability of metals to enter groundwater and the uptake of plants in the food chain. Some metals, such as zinc, copper and nickel are trace elements, and they play an important role in plant growth. Literature data (Saveyn and Eder, 2013) suggest that all types of composts contain PCB and PCDD/F, at least in trace amounts. However, the concentration range does not exceed the existing national limits.

10.3. SOIL MICROBIOLOGICAL PROPERTIES

Soil is the natural habitat for various microorganisms. Its main function is to supply plants with water and nutrients. Other soil functions include: maintaining biodiversity and creating a habitat for microbiological processes of matter circulation. Microorganisms perform a number of important functions, including: maintaining soil structure, releasing organic compounds, humification, utilization of pollutants and participating in the transformation of organic matter (Marcinkowska, 2002; Nannipieri et al., 2003; Paul and Clark, 2000; Preston et al., 2001). Soil microbiological properties are considered sensitive to changes in environmental conditions just as physical and chemical properties are. However, an additional advantage of monitoring microbiological properties is the possibility for faster observation of the effects of changes in environmental conditions. Qualitative changes in microorganism communities have a significant impact on the functional integrity of soil (Jezierska-Tys and Frąc, 2008). Soil microorganisms actively participate in the soil-forming process and together with the vegetation cover determine both the direction and nature of biochemical changes (Siebielec et al., 2015). Over 80% of all soil processes are closely related to the activity of microorganisms (Doran et al., 1996; Marinari et al., 2006). The degree of microbial development in soil depends on its physical and chemical properties, fertilization, climatic conditions and agrotechnical factors, and especially on the abundance of organic matter, which is a source of energy and nutrients for microorganisms (Johansson, 1999; Myśków, 1986). The soil consists of many different components, but organic matter is the most important among them. Soil organic matter (Fig. 10.1):

- creates soil structure and retains water,
- stores the necessary nutrients for plants,
- is the substrate and source of nutrients for microorganisms,
- binds organic coal.

Soil productivity and human ability to produce food depend on organic matter (Giacomazzi, 2019).

Cultivated soils rich in organic matter are characterized by higher biological activity. The use of microbiological indicators in soil environment analysis helps assess the ecological state of soils, their biological activity as well as fertility (Quemeda and Menacho, 2001). Knowledge of soil microorganisms together with other biochemical factors, makes it easier to understand the direction of processes occurring in soil and its current state. One of the indicators determining the quality, fertility and biological activity of soils is the ratio of the number of bacteria (including actinomycetes) to the number of fungi (Skwaryło-Bednarz, 2008; Wyszkowska, 2002).

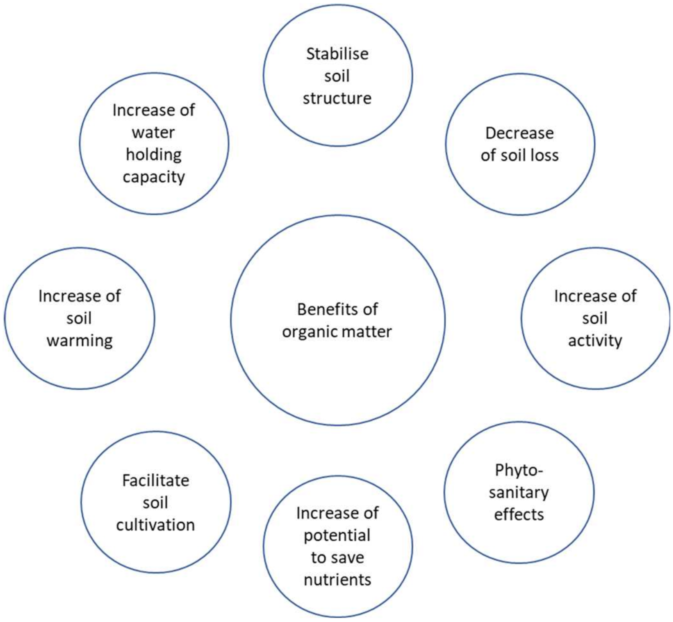

Fig. 10.1. Functions of soil organic matter

Larger values of this indicator inform us about weaker development of fungi, while smaller ones tell us about a stronger fungi position which is unfavorable from the point of view of soil fertility. Mineralization of organic matter is a process involving bacteria and fungi that provides the ingredients necessary for plant development. As a result of biological processes occurring in soil with the participation of microorganisms, soluble organic compounds are formed, such as simple sugars, simple organic acids, or amino acids (Bednarek et al., 2004). After the introduction of organic matter into soil the activity of soil microorganisms increases (Bastida et al, 2007; Fijałkowski and Kacprzak, 2009) and their presence promotes the release of nitrogen (Fijałkowski and Kacprzak, 2009). It is therefore justified to use microbiological and biochemical properties as indicators of soil quality because these parameters play a major role in the carbon and nitrogen circulation in the environment (Garcia-Gil et al., 2002; Janvier et al., 2007; Nannipieri et al., 2003). These indicators also define the metabolic processes of microorganisms: the intensity of nitrification, ammonification, fiber breakdown, enzymatic activity, and the production by microorganisms of specific metabolites or release of CO_2 (Janvier et al., 2007).

Sewage sludge is a rich source of nitrogen, its use as a fertilizer affects the microbial transformation of soil nitrogen and at the same time the number of ammonifying, nitrifying and denitrifying bacteria. Ammonification and nitrification processes inform us about soil nitrogen transformations (Jezierska-Tys and Frąc, 2008). These processes are also considered as indicators of soil biological activity and can be used to determine the impact of various factors on the biological state of the soil environment.

10.4. RELATIONSHIP BETWEEN THE PROPERTIES OF SEWAGE SLUDGE / BIOWASTE INTRODUCED INTO SOILS AND THE LEVEL OF SOIL MICROBIAL ACTIVITY

Soil microbiological activity is the sum of the metabolic activity of soil microorganisms: bacteria, actinomycetes, fungi, algae, protozoa, and invertebrate animals. Changes in soil microbiological activity are one of the measures of soil environment pollution (Encyclopedia). Sewage sludge is a slowly degradable organic material. The rate of this distribution depends on the C:N ratio, which is primarily important for microorganisms that easily use both components (Czekała, 2002; Garcia-Gil, 2002). Soil microorganisms, by transforming huge amounts of organic and mineral compounds, enrich the soil with nitrogen, growth substances, antibiotics, and biologically active substances (Corstanje and Reddy, 2006; Emmerling, 2002; Janvier et al., 2007). According to Jezierska-Tys and Frąc (2005) sewage sludge from the dairy caused a significant increase in the number of bacteria in soil. In addition, it was found that straw introduced into the soil with sludge also significantly stimulated bacterial growth, which was likely influenced by the C:N ratio. Organic matter introduced into the soil with sewage sludge has a direct impact on the overall number of bacteria and soil fungi (Karaca, 2002). The positive influence of sewage sludge on soil bacteria is confirmed by studies of many authors (Joniec and Furczak, 2007; Kacprzak and Stańczyk-Mazanek 2003; Nowak et al., 2010). Apparently, an increase in the number of bacteria, mainly oligo- and macrotrophic, as well as filamentous and cellulolytic fungi is observed in the layer of direct application of sludge (Joniec and Furczak, 2007). The number of bacteria and soil fungi is related to the level of organic carbon and the C:N ratio, occurring in the soil after the introduction of sewage sludge or bio-waste (Jezierska-Tys and Frąc, 2005; Marschner et al., 2003). In some studies (Loc and Greinert, 2000), a significant proportional increase in the number of tested microorganisms was observed along with the increasing dose of sludge, and the introduced organic matter also had a direct impact on the growth and yielding of plants and the improvement of the physical, chemical and microbiological state of the soil.

The application of sludge to the soil affects not only the growth of bacteria, but also fungi. Nowak et al. (2010) showed that sewage sludge increases the population of fungi in soils, such as Penicillium, Verlicillium, Mucor, Mortierelta, Fusarium, Geotrichum and Trichoderma. Other beneficial changes such as increasing pH and humidity improving air-water conditions could also contribute to the increase in the number of microorganisms, especially bacteria (Jezierska-Tys and Frąc, 2008).

In recent years, more and more research works concern soil fertility and biological parameters after its supplementation with organic waste (Andres, 1999; Frąc and Jezierska-Tys, 2011; Lopez-Mosquera et al., 2000;). Ros (2003) showed that along with the increase in the dosage of fresh and composted municipal waste, an increase in soil microbial activity was found. The use of sludge for fertilizing purposes affected the rapid growth of organic matter. The structure of microorganism communities is also rapidly changing after sludge application. The genetic profile of bacteria in soil after applying sludge differed significantly from the control object. The differences were noticeable even after 3 months from the application of sludge. A similar condition was observed for mushroom communities (Suhadolc, 2010). Kobus et al. (1988) determined the total number of bacteria and fungi in degraded soils fertilized with sewage sludge and compost. They showed that the number of selected microbiological indicators in soil depends on the dose of fertilizer used. Zielińska et al. (2012) found that the organic substance introduced into the soil together with compost "Dano" has a positive effect on the biological activity of soils, expressed by the increased number of ecophysiological groups of microorganisms. Microbiological analysis also showed that compost used in the experiment stimulated the microbiological activity of the soil. Korniłłowicz-Kowalska and Bohacz (2005) studied the effect of composts made of chicken feathers and pine bark as well as pine bark and rye straw on the microbiological, biochemical, and chemical properties of soils. In order to determine the total number of bacteria, fungi and cellulolytic microorganisms, soil samples were taken from wheat grown after cereals and legumes. The authors showed that the addition of compost activated the development of various physiological groups of microorganisms in the soil, while the intensity of their growth and development varied depending on the chemical composition and pH of the introduced compost.

Table 10.1

Applications and the effectiveness of methods of introducing sewage sludge and biowaste into soils, and the level of microbial activity of soils

Applications	Efficiency
The use of microbiological indicators in the analysis of the soil environment (helps to assess the ecological condition of soils, their biological activity and fertility) (Jezierska-Tys and Frąc, 2005a; Quemeda and Menacho, 2001)	The degree of microbial development in the soil depends on its physical and chemical properties, fertilization, climatic conditions, agrotechnical factors and content of organic matter, which is a source of energy and nutrients for microorganisms (Johansson, 1999; Myśków, 1986)
The use of microbiological and biochemical properties as indicators of soil quality after the introduction of sewage sludge (Garcia-Gil, 2002; Janvier et al., 2007; Karaca, 2002; Nannipieri et al., 2003)	Cultivated soils, rich in organic matter are characterized by a higher biological activity than soils, poor in organic matter
Increasing the yield and improving the physical, chemical and microbiological condition of the soil by introducing organic matter into the soil (supplied from sewage sludge and biowaste) (Janvier et al., 2007; Loc and Greinart, 2000; Zahir et al., 2001)	
Soil enzyme activity can be considered as an indicator of overall microbial activity (Bastida et al., 2007; Frąc and Jezierska-Tys, 2011; Jezierska-Tys and Frąc, 2008; Jezierska-Tys and Frąc, 2005a; Nannipieri et al., 2003)	

The effect of compost on the soil microorganism population is the sum of many factors that regulate the composition and activity of microflora (Nowak, 2001). The ways compost affects soil are quite complex, and it will take a long time to fully understand them. However, there is an agreement that the long-term effects of compost is beneficial for soil fertility. The applications and effectiveness of sewage sludge and biowaste introduction into soil related to the level of microbiological activity are given in Table 10.1.

Table 10.2

Strengths and weaknesses of the introduction of sewage sludge and biowaste to soil, and the level of microbial activity of soils. Advantages and limitations of the method

Strengths	Weaknesses
– High correlation between the amount of sewage sludge dose and soil enzymatic activity. – Organic matter brought into the soil with sewage sludge and biowaste has a direct impact on the increase in the total number of soil bacteria and fungi. – Sewage sludge is an organic material that decomposes slowly. The rate of this decomposition depends on the C:N ratio, which is important primarily for microorganisms that easily use both components.	– Soil contamination with heavy metals from sewage sludge reduces its enzymatic activity. – Higher value of the ratio of the number of bacteria and actinomycetes to fungi in the soil informs about a lower development of fungi, and lower values – about their stronger development, which is unfavourable from the point of view of soil fertility.
Method advantages	**Method limitations**
– Biological parameters, i.e., respiratory and enzymatic activity, are more sensitive and describe the state of the soil environment better than the physicochemical properties, as they are directly related to the microorganisms carrying out these processes. – The number of soil bacteria and fungi is significantly related to the level of organic carbon and the C:N ratio in the soil after the introduction of sewage sludge. – Positive influence of sewage sludge on the development of soil bacteria. – Measurement of soil enzyme activity can be used to better understand environmental disturbance caused by the functioning of the ecosystem.	– The enzymatic activity of soil fertilized with sewage sludge is related to the level of mineral (heavy metals) and organic (WWA, PCB) pollutants.

Niekerk and Claassens (2005) studied the intensity of the ammonification and nitrification process in soils fertilized with sewage sludge. Studies showed a positive effect on the intensity of these processes. In addition, the intensity of the ammonification process decreases with the time of soil incubation, while the intensity of the nitrification process increases (Niekerk and Claassens, 2005). Hernandez et al. (2002) observed significant differences in the intensity of nitrogen mineralization depending on the type of soil fertilized with sewage sludge. However, Speir et al. (2003) found no effects of sewage sludge on nitrogen mineralization and nitrification in studies. The number of ammonifying and nitrifying bacteria may inform us about the processes they carry out, although there is not always a correlation between the number of these microorganisms and the intensity of the ammonification or nitrification process, as stated by Jezierska-Tys et al. (2005a). Ammonification bacteria occur in sewage sludge and play a large role in the decomposition of organic nitrogen connections (Loc and Piontek, 2000). Loc and Greinart (2000) observed a significant increase in the number of ammonification bacteria under the influence of increasing doses of sewage sludge introduced into the soil. Nitrifying bacteria are key microorganisms involved in the regeneration of nitrates(V) because they are responsible for the oxidation of ammonia to nitrates(III) and then nitrates(V) (Corstanje and Reddy, 2006). Research by Gostkowska et al. (2000) showed sludge significantly increases the number of nitrifying bacteria in soil. The research by Loc and Greinart (2000) proved that the amount of sewage sludge introduced into the soil increased the number of nitrifying agents in it. Table 10.2 presents the technological strengths and weaknesses of sewage sludge and biowaste introduction into the soil and the resulting limitations and advantages of this method.

10.5. SUMMARY

Many factors influence the activity of microorganisms in soil environment. The availability of organic matter is associated with the reduction or stimulation of microbial activity. Soil fertility is shaped by the intensity of biological processes, which is related to the number of microorganisms. After introducing sewage sludge or biowaste into the soil they become a source of available organic carbon and macroelements. They stimulate the growth of microorganisms and enzymatic activity. Sewage sludge and biowaste, and more precisely the organic matter, have a positive effect on both the physical, chemical, and microbiological properties of soils (Jezierska-Tys and Frąc, 2008; Krzywy et al., 2008; Nannipieri et al., 2003). However, the use of sewage sludge carries the risk of microbial contamination. Previous observations showed that the majority of pathogenic organisms do not multiply in the sludge and gradually die after adding sludge to soil. Organic components introduced into the soil after some time cause a significant increase in soil pH and an increase in the number of yeasts and molds, and then contribute to an increase in the content of nitrogen and the total number of bacteria in the soil. For example, dairy sewage sludge introduced into the soil activates microbial populations depending on the dose of waste introduced into the soil, the time of its impact, as well as the type of soil (Jezierska-Tys and Frąc, 2008). Stimulation of the development of microorganism groups is most clearly visible in the presence of higher doses of waste (50 and 100 Mg/ha) (Jezierska-Tys and Frąc, 2008). The analysis shows that the intensity of development of individual groups of microorganisms is dependent on the dose of sludge introduced into the soil and the type of soil.

10.6. REFERENCES

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Chapter 11

Identification of the main limitations related to the agricultural use of products from biowaste and products produced on their basis, the impact of processing technology on the occurrence of contaminants

Beata KARWOWSKA

11.1. INTRODUCTION

Increasing global urbanization, industrial development, economic growth and a rising human population coupled with changing production and consumption types have a direct effect in the generation of huge amounts of diverse solid waste. Proper management of solid waste is one of the key tasks of the twenty-first century and a fundamental element for sustainable development (Lohri et al., 2017).

The basic biowaste related legislation includes Directive 86/278/ECC (Directive, 1986) regulating the application of sewage sludge for agricultural purposes. Its role was to prevent a soil ecosystem to minimize the harmful effect of hazard sludge components. The limited values relate to the heavy metals content in applicate sludge as well as in amended soil and additionally, a maximal annual content of heavy metals that would be introduced to the soil.

Another key legal regulation is formulated in the Landfill Directive (Directive, 1999) determining the percentage of biodegradable wastes diverted from landfills compared to the percentage generated in 1995. The directive does not prescribe the methods or options for wastes treatment and, as a result, European Union Member States often opt for for the cheapest and easiest methods for waste dispreading without consideration of future environmental consequences (Estrada de Luis et al., 2013).

The basic recommendation for biowaste management in United Europe in a formal legislative act was stated in the Waste Framework Directive (Directive, 2008). The legal act determined the idea of biowaste and recommended selective collection of wastes for future treatment and sufficient application. It provides

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requirements and sets the basic waste management definitions for the EU Members. The Directive required the assessment of biowaste management. The promoted activity in biowaste management should promote (Estrada de Luis et al., 2013):

- separate collection and recycling generally oriented to composting or anaerobic digestion,
- biowaste treatment to assure a high level of environmental protection;
- the use of safe products of biowaste transformation.

In 2018 innovation in a form of a circular economy package was enforced. It included legislative amendments to the Waste Directives. The most important changes were (Jędrczak, 2018):

- increase in the ratio of recycled municipal waste to 55, 60 and 65% by 2025, 2030 and 2035, respectively;
- increase in the ratio of recycled package materials to 65 and 70% by 2025 and 2035, respectively;
- obligatory selective collection of textiles and hazardous waste from the 1st of January 2025;
- decrease the level of disposed municipal waste to 10% by 2035;
- support in economic instruments that discourage waste storage;
- prohibition of storage of segregated waste;
- recommendation of biowaste segregation and recycling at the source or separate collection without mixing with other types of waste;
- including biowaste to the specific type of waste, that after transformation could cease to be waste;
- encouraging a reduction in food waste in 30 and 50% by 2025 and 2030, respectively.

From the point of view of application, it is very important to know the definition of biowaste, biodegradable waste and other terms. According to the European Union Waste Framework Directive (Directive, 2008), biowaste is defined as biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and also includes comparable waste from food processing plants. We have to remember that it does not include forestry or agricultural residues, manure, sewage sludge or other biodegradable waste such as natural textiles, paper, or processed wood. Biowaste should not be confused with the wider term "biodegradable waste" as defined in the Landfill Directive (Directive, 1999), which also covers other materials capable of undergoing anaerobic or aerobic decomposition as food, garden waste, paper, cardboard, sewage sludge, natural textiles. In the literature nations: "biowaste", "biodegradable waste" and additionally "biosolids" are used interchangeably, and it is sometimes difficult to detect the differences between them (Estrada de Luis et al., 2013; Horsák et al., 2014; Lohri et al., 2017; Sharma et al., 2017).

From the point of view of waste management, a few other definitions should be clarified. The first of them is "compost". Compost is generally the solid material produced during composting process. The composting process produces sanitized and stabilized product, practically ready to future reusing. Another notion is "sewage sludge", material which is not detected as a biowaste in the Waste Framework Directive but is a very important substance for next nature utilization. Sewage sludge is defined as a residual sludge from sewage plants treating domestic or municipal waters (Directive, 1986). And "digestate" is defined as a semi-solid or semi-liquid product of anaerobic digestion of biodegradable materials.

The goal of the presented chapter was to present a short introduction for biowaste management as well as short overview of the methods for biosolids transformation before agriculture reusing. The general advantages and disadvantages as the limitations for potential natural application were presented in the context of didactic and other educational purposes.

11.2. BIOWASTE MANAGEMENT

The general environmental impact form biowaste as well as other biodegradable waste is the formation of methane that takes place during their decomposition in landfills. Releasing of methane to the atmosphere has significant contribution to the greenhouse gasses content. The reduction of this problem would be reached as a result of decreasing the amount of biodegradable municipal waste that is stored in landfills. The Landfill Directive does not indicate concrete treatment options for the waste. The most important benefit is good biowaste management and limitations in greenhouse gases emissions. Additionally, the formation of good quality compost for the improvement of soil quality and the production biogas are beneficial to resource efficiency and provide energy self-sufficiency. In practice, the easiest and cheapest options are chosen such as incineration or landfilling. The Directive assumes some waste management principles (Estrada de Luis et al., 2013). It requires that waste be managed without:

- endangering human health,
- harming the environment, and in particular,
- risk to water, air, soil, plants or animals,
- causing a nuisance through noise or odors,
- negatively affecting the countryside or places of special interest.

The hierarchy of working with substances in agreement with the Waste Framework Directive is presented on Figure 11.1. Potential waste management includes several levels like – disposal, recovery, recycle, reuse and prevention. Only the main product is retained with prevention of its distinctive properties. Every form of waste should be reused, recycled or recovered. The storage or disposal in the environment is the last possibility (Estrada de Luis et al., 2013; Sharma et al., 2017). This scheme would be also suitable for managing biowaste.

An agriculture recycling of biowaste is the environmentally preferred option, making it possible to maintain valuable nutrient elements for plants. Because of the presence of organic matter, nitrogen and phosphorus compounds in organic and inorganic fractions, biosolids can work as soil amending agents. The levels of biowaste formation and various possibilities of their disposal are summarized in Table 11.1 for selected developed countries (Sharma et al., 2017).

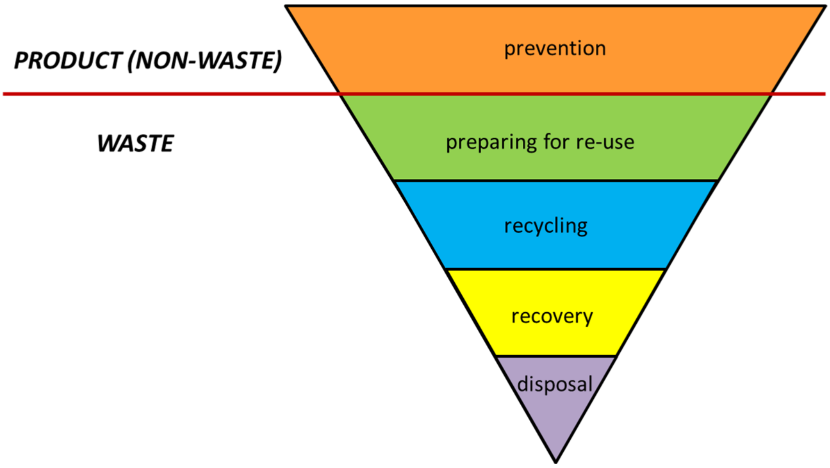

Fig. 11.1. Scheme of working with substances in the context of future application

Table 11.1

Biowaste production and utilization with different methods in selected developed countries

Country		USA	United Kingdom	Australia	Japan	Germany
Biowaste production 10^6 Mg d.m./year		17.80	1.05	0.36	2.20	2.30
% utilization		55	85	80	74	60
% of method	agriculture	36	79	55	12	30
	landfill	29	1	5	13	3
	thermal disposal	15	18	3	75	53
	other	20	2	37	0	14

d.m. – dry matter

Apart from advantageous properties, waste material can contain toxic heavy metals, or other undesirable elements from the point of view of land application. Limits regarding allowable concentrations of heavy metals in different waste origin substances used for agricultural purposes in the European Union are presented in Table 11.2 (Estrada de Luis et al., 2013). Increase in heavy metal content is usually the result of inappropriate technology of waste treatment and may cause a serious environmental risk of soil pollution.

Heavy metal content in compost used for land application is limited in different countries around the world. Examples of limitations are shown in Table 11.3 (Sharma et al., 2017).

Besides heavy metal, waste material ions could be contaminated with organic micropollutants like dioxins, polychlorinated biphenyls, polyaromatic hydrocarbons, pesticides, insecticides, cosmetics, pharmaceuticals, detergents, hormones as well as different types of inorganic salts (Estrada de Luis et al., 2013; Sharma et al., 2017). Long term land application of biowaste can cause the accumulation of hazardous or toxic substances that could transfer to the food chain in diverse ecosystems. Nevertheless, in accordance with actual legislative acts, determination of organic toxic substances level is not necessary before land deposition of biowastes or their treatment by-products. Donation of organic matter to soil can improve the physical properties of soil like: increasing the water holding capacity, formation of stable fraction of heavy metal ions and decreasing metal availability to plants.

Table 11.2

Heavy metal limits for sludge, biowaste and compost

Waste type	Metal content, mg/kg d.m.						
	Zn	Cu	Ni	Cd	Pb	Cr	Hg
sewage sludge	2500-4000	1000-1750	300-400	20-40	750-1200	–	16-25
compost/organic farms	200	70	25	0.7	45	70/ 0(Cr(VI))	0.4
compost eco-label	300	500	50	1	100	100	1
stabilized biowaste	800	500	100	3	200	300	3

Table 11.3

Limits of selected heavy metals content in compost used in land application (Sharma et al., 2017)

Country	Metal content, mg/kg d.m.						
	Zn	Cu	Ni	Cd	Pb	Cr	Hg
USA	400	300	50	4	150	100	0.5
Canada	500	100	62	3	150	210	0.8
Australia	250	200	-	3	200	–	–
India	1000	300	50	5	100	50	0.15

Biowaste treatment technologies are processes that change discarded biowaste into new and potentially valuable products. Technologies of biowaste treatment or management could be grouped into four basic categories: direct use, biological treatment, physico-chemical treatment, and thermochemical treatment. The overview of the mentioned processes is presented on Figure 11.2 (Lohri et al., 2017; Sharma et al., 2017). Products of some processes as well as biowaste can be used for agricultural purposes. In this context the most important seem to be direct application and biological treatment.

The direct use of biowaste is the common form of waste application. It is characterized by simple operation and low cost but is technologically outdated (Lohri et al., 2017). Three general types of activity could be mentioned in this category:

- direct land application (landspreading) – raw waste disposal onto fields, in practice the spread of raw agricultural wastes (manure, crop residue, etc.);
- direct feed for animals – recovery of valuable nutrient components of waste;
- direct combustion – environmentally dangerous with the possibility of uncontrolled emission of hazards.

The risk of such application of biowastes depends on their composition. The pollution present in them can freely affect environmental condition and, as a result, human and animal health. It is historically a group of the oldest and inefficient methods, but it is still used in practice, especially in rural settings.

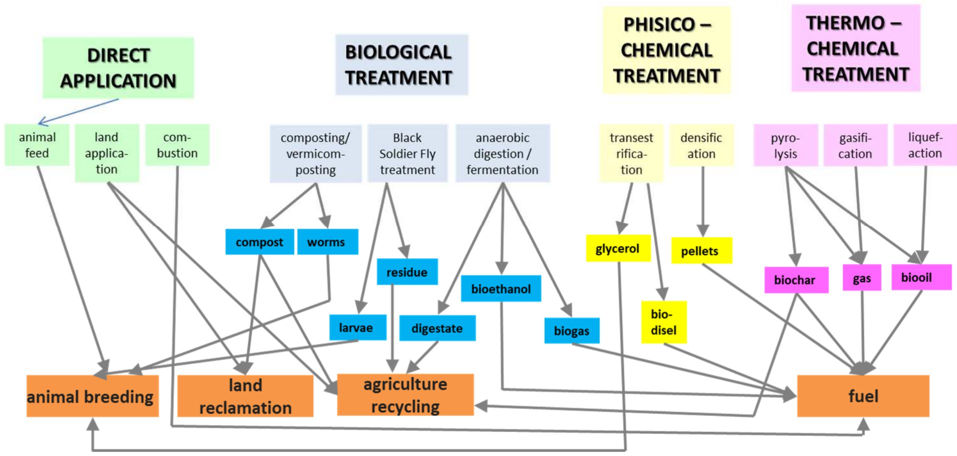

Fig. 11.2. Scheme of solid biowaste management processes (Lohri et al., 2017; Sharma et al., 2017)

Apart from the direct disposal of biowaste, for agriculture purposes, the products of biological transformations of waste seem to be a more efficient way to manage them. The most useful processes are (Horsák et al., 2014; Lohri et al., 2017; Sharma et al., 2017):

- composting (or vermicomposting) – decomposition of organic components of waste under anaerobic conditions with formation of a stable organic final product named humus; when additional interactions of earthworms are involved, the product of their feeding on waste is earthworm castings, called vermicompost;
- Black Soldier Fly (BSF) treatment – relies on transformation of organic biomass by existing larvae of a domestic fly with the main products: larvae and residue;

- anaerobic digestion (AD) – decomposition of complex organic matter under anoxic conditions to methane and carbon dioxide with formation of useful products named: digestate, biogas and bioethanol.

Biological processes are controlled transformation of waste using living organisms. Biochemical and biotechnological conversion are usually applicable for materials with significantly high moisture content as living organisms require water for their metabolism.

11.3. POTENTIAL ENVIRONMENTAL RISK OF DIFFERENT METHODS OF BIOWASTES APPLICATION

After direct land disposal of biowastes, a raw organic part of them undergoes a natural degradation process under aerobic conditions. As a result of biodecomposition, organic matter and nutrients are transported to the soil. The overall shape of the process should improve the soil properties. Degradation of organic matter of biowaste may additionally cause nitrogen competition in soil: microorganisms, for their own metabolism, compete the plant in the demand of nitrogen. As a result, the cultivated plants show symptoms of nitrogen deficiency (Smith et al., 2015).

Raw biowaste is nutrient elements rich material, and presence of it may result in mobilization and leaching of nutrients into surface and ground water or in volatilization of ammonia (Lohri et al., 2017).

Direct land application is not a treatment process and could affect the quality of the soil or plant condition. A waste can contain trace elements like heavy metals, which are toxic for organisms in any element of the environment, and because of their presence in soil, they could enter into the food chain after uptake by plants, through animal organism to the human body (Chen et al., 2016). Unlike organic pollution that can be biodegraded with time or can be combusted, metals are not degradable, accumulate in living organisms, and remain a real potential threat to both the environment and human health (Babel et al., 2006; Udom et al., 2004).

Additionally some literature reports indicate the risk of lower availability of micro-nutrients for plants growing after addition of no stabilized organic substance. Land disposal of raw biowaste must be restricted and controlled to protect environmental risk.

The use of untreated biowaste for soil amending has another potential danger. The raw organic material could contain pathogens, and direct disposal does not guarantee neutralization or removal of them. In the European Union, the possibility of wastes spread on land needs preventive physical and chemical analysis of waste as well as soil before any direct application of raw wastes (Lohri et al., 2017).

Composting is one of the oldest methods of biowaste management. There is some documentation of ancient Greek and Roman, as well as early Asian and American civilizations regarding use for degradation of wastes (Jędrczak, 2018; Lohri et al., 2017). The wide types of solid wasting materials could be driven to composting: food residuals, grass, leaves, branches, agriculture wastes, manure, human feces and even municipal wastes. When the introducing material is poor in water, for initialization or driving process the addition of water is necessary to ensure activity of the microorganisms involved in the process. It especially depends on climate or structure of the feedstock substance.

The microbial population decomposes organic matter and as a result of biochemical processes, water, carbon dioxide and thermal energy are formed. Basic parameters ensuring the proper process occurrence are: C:N ratio, aeration, temperature, pH, moisture content, size of particles. Controlling them helps to achieve a fast process rate and assists in obtaining a good quality product. If the conditions are not in the optimal range, the speed of composting may be greatly reduced or even not occur at all. Composting proceeds by three steps:

- mesophilic phase (a couple of days);
- thermophilic phase (from several weeks to months) – temperature 55-70°C, hygienization of a subjected substance;
- cooling and maturation phase (several months) – temperature inside the pile comparable to ambient temperature.

The basic product of composting is similar to soil; a stable, dark – brown solid, with earthy smell and crumbly texture. Other outflowing products are leachate, water and carbon dioxide. Compost contains nutrient elements: nitrogen, phosphorus and potassium, macro- and microelements, microorganisms and is rich in humus important for plant cultivation. It could be applied to amend solid, soil remediation, for landfill cover and land restorations (Lohri et al., 2017). Through composting, biowastes could be transformed into valuable products with simultaneous reduction of the waste volume. Among the common applications of compost as a fertilizer or soil, reclamation material is a potential use of easily combusting compost for energy production (Chia et al., 2020; Chojnacka et al., 2020). Besides the many appreciable features, urban waste composting is not so widespread. The main reason could be a low quality of biowaste material which may result in weak quality of a final product. Additionally, requirements of the biotechnological process may result in the formation of odors and vermin that are harmful to the environment and may finally lead to no acceptance of inhabitants or users of compost (Jędrczak, 2018). What's more, insufficient government policies and marketing experience often affect the economy of the composting process. Another problem connected with use of compost is the potential for heavy metal occurrence.

The innovation of composting is vermicomposting process – an anaerobic process of organic matter decomposition with cooperation of microorganisms and earthworms under controlled conditions. Earthworms are able to exist on household waste, municipal organic waste, organic residues of food, wood, paper industries and even on sewage sludge (Cui et al., 2020; Garg et al., 2006; Padmavathiamma et al., 2008). Some types of food waste are inappropriate for earthworms. They do not accept meat, fish, dairy products, grease, vinegar, and table salt.

In vermicomposting, the composting process is preceded the initial pre-phase in which the microbial population prepares biowaste for earthworms' existence by aerobic initial degradation of waste. It improves the feeding process of worms. On the other hand, earthworms feed on waste product fecal material promoting the activity of microorganisms and increasing the quality of final vermi-compost material. The increased activity of microorganisms can increase temperature in biowaste layers or promote anaerobic conditions. Both situations are unfavorable for earthworms.

General physical and chemical parameter characteristics of the vermicomposting process are: pH, temperature, moisture content, feeding rate, C:N ratio.

Compared to compost, vermicompost is characterized by a finer texture and a slightly higher nutrient content that is available for plants. Technology reduces organic material, increases nutrient elements availability as a result of biological element fixation and solubilisation of phosphorus. Vermicompost stimulates the growth of plant roots, increases nutrient adsorption and finally aids in the efficiency of agriculture production (Padmavathiamma et al., 2008). The enrichment of soil with vermicompost has a significant influence on P, N and K content and, what's more, helps in the management of land without affecting the environmental processes. An additional advantage is the fact that heavy metals present in initial material accumulate in earthworms' bodies and could be much more easily removed from the final substance (Lohri et al., 2017). What's more, vermicomposting exchanges harmful components of waste to nontoxic forms.

The second basic product of vermicomposting are earthworms themselves. Worms are rich in protein with all key amino acids, so they can be applicable for the feeding of animals.

Vermicomposting is not widespread technology in waste treatment. There are some limitations to it:

- requirement of a large space;
- required initial pretreatment of material before earthworms feeding;
- low quality of introduced material;
- necessity of high attention of process parameters;
- knowledge of biochemical process requirements – adequate skills, understanding of lifecycle of earthworms, optimization of the process condition;
- political and economic disadvantages – poor favoring policy, limited marketing efforts, excessive expectations of revenues.

The innovative technology of organic biowaste transformation into insect oil or protein is method called Black Soldier Fly (BSF). Process uses natural appetite of some fly and their larvae for decomposition of organic matter. Wide types of biowaste material are suitable for BSF process. Larvae can exist on food (involving meat and fish) and market waste, animal manure and human excrements (Gold et al., 2020). The process occurs with biowaste with an adequate moisture content on the level of 65-80%. This requires either dewatering of wet waste either adding water to dry substrate. Another limiting parameter is the temperature: 25-32°C which is suitable for all of BSF periods of life. The BSF larvae can reduce the feedstock mass by 50-80% and convert up to 20% into larval biomass within about 14 days.

It is a technology with high economic potential, but it relies on the colonization of wastes by the natural fly population and is not useful for controlled waste disintegration (Lohri et al., 2017).

The substantial product of BSF waste treatment are fly larvae and residue, both with potential ability for agriculture application. High protein content indicates larvae from BSF process as a suitable animal feed for fish, pigs, or poultry in particular (Makkar et al., 2014; Lohri et al., 2017). The residue of the process still contains nutrient elements on a significant level, so it could be applied for soli reclamation. The same problem is connected with the short time of process and residue has to be driven to additional maturation step to avoid oxygen depletion in donated soil, which would inhibit plant germination and growth. What's more, land application of residue carries a serious risk of soil contamination with pathogens or toxic compounds like heavy metal ions, pharmaceuticals, and pesticides. The BSF process does not sanitize waste sufficiently but reduces the population of *Salmonella* spp. Other pathogens such as *Enterococcus* spp., helminth eggs or bacteriophages are practically not sensitive for the BSF treatment process. On the other hand, heavy metals present in biowaste may accumulate in bodies of larva and generate the needs of additional precautionary measures. The advantage of BSF is degradation of pharmaceuticals and pesticides.

Modification of BSF larvae feedstock material could result in rising of biowaste treatment performance and reduces the variability of the process. Preparation of modern biowaste mixtures, which contain protein and non-fibre carbohydrate content, seems to be a promising approach for an efficient and predictable BSF operation system for biosolids treatment (Gold et al., 2020a; Gold et al., 2020b).

The existing of living larvae (or other animals) in big masses requires a special regime of operation. Delivery of feedstock material and pick up of the product has to be adequately synchronized. BSF technology is suitable only in some regions: tropics and sub-tropics over the world.

Actual legal barriers limit the development of the BSF waste treatment method in several countries (Lohri et al., 2017).

Another biological method of biowaste treatment is anaerobic digestion. It is an efficient, well-defined, engineered process of biochemical decomposition of liquid and solid organic matter by various bacterial activities under oxygen free conditions (in closed reactors – digesters) at temperatures suitable for activity of mesophilic or thermophilic bacteria. The process occurs in nature in anoxic environments including: watercourses, soils, landfills, and in animal digestive systems (Lohri et al., 2017; Jędrczak, 2018).

A wide variety of wastes can be used as feedstock substance in the anaerobic digestion process: sewage sludge, animal manure, slaughterhouse and food industry waste, harvesting residues, energy crops as well as the organic fraction of municipal solid waste. Generally, the process requires an optimal level of water content. For this reason, material with high moisture can be conducted to degradation without additional pretreatment. From the point of view of substrate for the anaerobic digestion process, the main limitation is using ligneous organic substances, e.g. wood. Such material is not degraded by anaerobic microorganisms.

The characteristic factors for the anaerobic digestion process are pH, temperature, moisture content, C:N ratio, inoculation, stirring, retention time and substrate type. The AD process could be inhibited dependently on substances forming or existing in the surrounding environment. One of the parameters causing inhibition of the process is the excessive acidification and resulting stress and inhibition of methanogenic bacteria. Such a situation occurs when feedstock contains large amounts of easily biodegradable organic material, and the intensive production of volatile fatty acid is observed. Other inhibitors of anaerobic digestion are ammonia, sulfides, active metal and heavy metal ions (Chen et al., 2008; Cui et al., 2020).

Table 11.4

Summary of advantages and limits of biowaste transformation processes in the context of agriculture application

Biowaste transformation method	Characteristics	Advatages	Limits
Direct disposal (DD)	– the oldest inefficient method; – application without any initial treatment: field disposal, animal feed;	– simple operation; – low cost; – recovery of valuable nutrient elements;	– outdated technology; – pollution freely and directly affect environment and finally human and animal health; – pathogen organisms practically not sanitized; – necessity of preventive analysis of waste and soil; – nitrogen competition microorganisms vs. plants;
Composting (C)	– decomposition of organic components with final stable organic product with using of microorganisms;	– basic product similar to soil; – rich in humus; – significant content of nutrient elements; – wide types of solid biowastes for process; – additional valuable product: thermal energy;	– need to control a lot of parameters (C:N ratio, aeration, temperature, moisture content, size of particles); – possible content of heavy metals pollutants; – climate dependent; – formation of harmful odors and vermin;
Vermicom-posting (VC)	– an anaerobic process of organic components decomposition with using of microorganisms and earthworms under controlled condition;	– finer texture comparing to compost; – higher nutrient element content comparing to compost; – nutrients easy available for plants; – possibility of heavy metals removing from organic matter (accumulation in earthworm organisms); – changes harmful organic material to nontoxic form; – earthworms additional valuable product (animal feed);	– some unacceptable materials (meet, fish, dairy products, grease, vinegar, table salt); – requirement of much space; – high attention in controlling of process parameters; – necessary of knowledge in biochemical processes – highly educated stuff;

Cont. Table 11.4

Black Soldier Fly (BSF) treatment	– transformation of organic biomass by domestic fly larvae – presence of high moisture content;	– wide types of solid biowastes for process (including meat, fish, animal manure, human excrement); – products: residue (agriculture) and larvae (animal fish); – significant content of nutrient elements; – possible degradation of some pharmaceuticals;	– adequate moisture content in the range 65-85% (necessity of dewatering or adding water); – risk of soil contamination (pathogens, toxic compounds, heavy metals ions); – not useful for controlled waste decomposition (domestic fly); – does not sanitize waste;
Anaerobic digestion (AD)	decomposition of complex organic matter under anoxic condition	– valuable products: digestate, biogas, bioethanol); – transformation of waste material into renewable energy source; – nutrient elements content;	– optimal water content; – some unacceptable materials (ligneous materials); – need to control a lot of parameters (C:N ratio, pH, temperature, moisture content, stirring, substrate type, inoculation); – inhibitors: ammonia, sulfides, active metal ions, heavy metal ions; – does not sanitize waste completely (presence of some pathogen organisms)

The most important benefit of anaerobic digestion is the transformation of waste material into a renewable energy source while keeping the content of nutrient elements in the digestate. The formed digestate contains a significant level of nitrogen compounds dependent on the type of decomposed waste. The product can be used for agricultural purposes as a fertilizer or organic matter amendment. When a substance can be incorporated into soil, the limitations concern the presence of hazardous components. The process of anaerobic digestion does not totally inactivate bacteria, viruses, parasites and seeds of weed. Pathogen neutralization depends on temperature, pH, time of process and content of easily degradable organic compounds. The higher hygienization effect is reached when a process occurs at thermophilic temperature above 50°C and eventually after an additional step of digestate treatment, for example aerobic composting (Lohri et al., 2017).

Additional problems concern the operational system of process. The number of factors that have an influence on anaerobic digestion occurrence cause difficulty in the selection of technology or designing and construction of reactors for digestion. Inappropriate operation, poor ownership and operator responsibility, and week business models are the most important reasons of failure.

Other chemical and thermochemical processes are of less importance from the point of view of agriculture management. Only transesterification products like glycerol and pyrolysis biochar have potential application for animal feeding and soil amending, respectively (Angouria-Tsorochidou et al., 2021; Lohri et al., 2017).

The comparison of advantages and limits of presented biowaste transformation technologies are summarized in Table 11.4.

11.4. SUMMARY

Biowaste is a valuable source of organic matter and nutrients for plant growing. Such features make them an important alternative to expensive inorganic fertilizers.

The addition of treated biowastes to soil has been found to be beneficial to soil health, enriching soil with essential nutrient elements as material increasing the pH and improving physical properties of the soil. The solid waste recycling sector not only needs further technological development, but also improved operating standards, product standards and enhanced market development.

The agricultural application of biowaste has a positive effect on plant cultivation. However, the content of heavy metals, pathogen organisms and another organic and inorganic hazardous and toxic compounds is a substantial limitation for its land application and plant growth medium due to the associated threat of leaching and food chain contamination due to these substances.

A wide range of treatment methods for solid biowaste already exist and have been extensively studied. Each technology can use biowaste of a specific composition and origin whereby some technologies are more restrictive in their requirements than others. Conditions of method could be a serious barrier for their use. Each technology can be described by relevant process steps and parameters to generate products with different properties. Additionally, knowledge and experience requirements as well as policy of local or governmental systems seem to be basic problems in the application of useful or innovative technology of waste management.

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Chapter 12

Influences of sewage sludge as a biowaste on the development of phytopathogens

Dorota NOWAK

12.1. INTRODUCTION

In a world of limited resources of the biosphere and progressive climate change, only harmonious human-environment cooperation guarantees the preservation of the biodiversity of ecosystems, and thus their biological balance. Global population growth and an increasing amount of goods produced inevitably leads to an increase in the amount of waste, including sewage sludge. Sewage sludge is sludge released at individual stages of sewage treatment and comes from fermentation chambers and other installations used for municipal sewage treatment. The properties of sewage sludge depend on the type of treated sewage, treatment technology and methods of stabilization. The growing requirements for environmental protection and the introduction of more and more effective methods of wastewater treatment generate an increasing amount of sludge requiring proper management. These wastes, known for their fertilizing qualities, are often used in, inter alia, soil feeding or are converted into compost. Properly selected and properly applied doses of sediment to the soil can improve its structure and increase the content of organic carbon and nutrients important for plants, such as nitrogen, phosphorus, and potassium. The fertilizing value of sewage sludge is comparable in this respect with manure. Skilful application of this waste can stimulate the microbiological activity of soils and positively affect both the growth of biomass and the number of soil microorganisms. It was found that in samples of soils fertilized with sewage sludge there was a significant increase in the activity of dehydrogenases and enzymes catalyzing cellular respiratory processes (Vieria et al., 2003; Placek et al., 2014).

12.2. SOIL – THE RICHEST LIVING ENVIRONMENT

Soil as the most biodiverse environment provides:

- production of plant biomass, ie food for the entire biosphere
- water retention
- production of humus
- contamination retention and decomposition

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The proper functioning of the soil is primarily determined by biological diversity based on the species richness of bacteria, actinomycetes, fungi, protozoa and plants and animals. Microorganisms play a special role in the transformation of organic matter, thus supplying plants with minerals. The proper functioning of the soil is primarily determined by biological diversity based on the species richness of bacteria, actinomycetes, fungi, protozoa and plants and animals. Microorganisms play a special role in the transformation of organic matter, thus supplying plants with minerals. As a result of microbiological processes, humus is created, the content of which determines the proper structure of the soil. Symbiotic bacteria capable of assimilating free nitrogen from the air and supplying it to plants as well as specific interactions between plant roots and mycorrhizal fungi play an extremely important role.

Mycorrhiza is a phenomenon of cooperation between fungi and tree roots (Fig. 12.1), which enables plants to receive water with mineral salts and fungi to provide them with the necessary assimilates. In addition, mycorrhizal fungi constitute a natural protection of plants against pathogenic organisms, i.e. phytopathogens (Chen et al., 2018).

Fig. 12.1. Mycorrhizal symbiosis between Hebeloma mesophaeum and the root of a spruce (https://www.cleantechloops.com/mycorrhizae/)

Therefore, the natural and agricultural use of sewage sludge must be carried out in a controlled manner, so that the proper functioning of agroecosystems is not disturbed. The agroecosystem consists of biocenoses shaped and regulated by man, the species composition of which is modified by economic needs (Fig. 12.2). The main consumer in agroecosystems is humans or livestock. The main role of agrocenoses is to provide agricultural crops to meet food needs and forage. It is related to the intensification of production and the use of artificial fertilizers and pesticides (Fig. 12.3). The use of sewage sludge as substrates for soil fertilization may contribute to the reduction of both artificial fertilizers and chemical plant protection products.

Fig. 12.2. Agroecosystem (https://pl.radiopachone.org)

Fig. 12.3. The use of pesticides in agrosystems (https://www.merieuxnutrisciences.com/pl)

12.3. SEWAGE SLUDGE – PROPERTIES AND THREATS

Sewage sludge (Fig. 12.4) as by-products of sewage treatment is heterogeneous in terms of physical, chemical, and biological properties. In addition to valuable fertilizer ingredients, they are characterized by high hydration, a significant content of organic matter and the presence of toxic chemicals such as: heavy metals, PAHs, PCBs (Dąbrowska and Rosińska, 2011) and a diverse biological composition.

Fig. 12.4. Sewage sludge (https://sozosfera.pl)

The following types of sludge are distinguished:

- primary sludge formed during sedimentation of easily falling suspensions in the primary settling tank (A),
- secondary sludge separated in secondary sedimentation tanks after biological wastewater treatment (B),
- mixed sludge formed after mixing primary and secondary sludge (C).

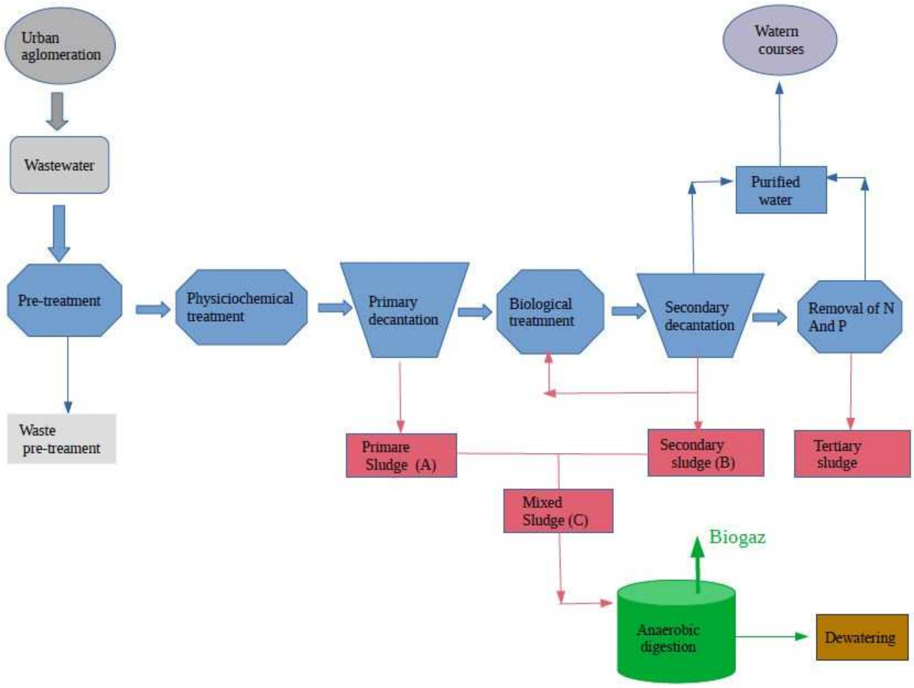

Fig. 12.5. Processes and technologies for sludge treatment

Due to the high content of organic matter, high hydration, and the presence of pathogens, sewage sludge must be treated. The treatment of sewage sludge is aimed at stabilization, effective dewatering, and improvement of the sanitary condition of the sludge, which enables its further management. Sediment stabilization is a process in which organic matter is transformed into inorganic matter. The stabilization processes are carried out in dedicated devices and enable the reduction of organic matter content by at least 38%. The stabilization of sludge can be carried out with the use of biological processes such as methane fermentation, composting, or chemical processes, e.g. liming or thermal processes. The most frequently used method is methane fermentation (Fig. 12.5), which makes it possible to reduce the mass of sludge by 40%, reduce odor nuisance, increase susceptibility to dehydration and partially reduce pathogenic organisms. The biogas produced during the process, containing approx. 60-70% methane, is a valuable energy resource that is used for the needs of the sewage treatment plant.

As numerous studies show, not only sludge directly separated from sewage, but also subjected to stabilization processes, is a habitat for viruses, bacteria, fungi, and helminth eggs (Pepper et al., 2006). The availability of easily degradable organic matter in sediments creates favorable conditions for the survival of these organisms, and the most commonly used methods of stabilization do not provide a sanitary safe product. Among the detected organisms, we can distinguish both saprobionts and relative or absolute pathogens, including plant pathogens, the so-called phytopathogens. These organisms, reaching the soil with sediment, can spread to plants, cause diseases and, as a result, destroy entire crops.

12.4. EXAMPLES OF DISEASES CAUSED BY PHYTOPATHOGENS

Phytopathogens, i.e. biotic infectious agents, include, first of all: viruses, bacteria and fungi that infect the plant through spores, conidia, mycelial fragments, or a vegetative cell. It is estimated that during the year, due to the contamination of crops with phytopathogens, a decrease in food production may reach 50% (Liu et al., 2013; Sharma et al., 2009). Viruses that cause plant diseases are:

- barley yellow dwarfism virus,
- potato streaky virus Y,
- cauliflower mosaic virus,
- brown spotted tomato virus.

Examples of diseases caused by viruses are shown in Figures 12.6 and 12.7.

Fig. 12.6. Yellow dwarfism of barley (https://www.fwi.co.uk/arable/crop-management/disease-management/how-better-aphid-tracking-could-reduce-barley-yellow-dwarf-virus)

Fig.12.7. Brown spots of tomatoes (https://www.google.com/search?q=Brown+spot+of+tomatoes)

All viral diseases cause inhibition of plant growth and their dwarfism, numerous changes in leaf color and necrosis of whole plants (Scholthof et al., 2011).

Among the bacterial diseases should be mentioned:

- wet potato rot caused by pectinolytic bacteria of the genus *Pseudomonas*,
- wet root rot caused by *Pectobacterium* and Dickeya bacteria,
- plant cancer caused by *Pseudomonas syringae*,
- the black rot caused by *Xantomonas campestris*.

Examples of diseases caused by bacteria are shown in Figures 12.8 and 12.9.

Fig. 12.8. Wet potato rot caused by pectinolytic bacteria the genus *Pseudomonas* (https://www.google.com/search)

Fig.12.9. The black rot caused by *Xantomonas campestr* (https://hort.extension.wisc.edu/articles/black-rot-crucifers)

The effects of infection by bacterial phytopathogens are wilted leaves, a blackening base of the shoot, and the absence of daughter tubers. Bacterial phytopathogens develop not only during vegetation, but also during the storage of vegetables, and the enzymes secreted by them digesting the cell wall of plants facilitate the penetration of viruses. Diseases caused by fungi are particularly burdensome for crops. Currently, more than 10,000 species of fungi that can cause plant mycoses have been classified, the most serious of which are, among others, representatives of the genera: *Penicillium*, *Botrytis*, *Monilinia*, *Rhizopus*, *Alternaria*, *Aspergillus*, *Fusarium*, *Geotrichum*, *Gloeosporium* and *Mucor* (Barkai-Golan, 2001). Among fungal plant diseases, fusariosis deserves special attention, as it is considered the most dangerous and the most phytopathogenic and phytotoxic in terms of destroying barley, maize, wheat, and rye crops (Dean et al., 2012, Wiśniewska et al., 2014). They infect plants in all development stages and, after the growing season, they can remain in the field on crop residues. Fusarium mushrooms are well adapted to changing weather and soil conditions. They are also characterized by high variability and significant tolerance to active substances used in plant protection products, which makes it difficult to select effective preparations. Many of these types of phytopathogens also have the ability to produce specific metabolites, the so-called mycotoxins that pose a serious threat to human and animal health and life (Chełkowski, 2010). Apart from nitrosamines, mycotoxins are classified as particularly dangerous environmental poisons, characterized by a strong cytotoxic, mutagenic, and carcinogenic effect (Nesić et al., 2014). The most important mycotoxins both economically and toxicologically on a European and global scale are:

- aflatoxin B1, ochratoxin A produced by *Aspergillus*,
- aflatoxin B1, ochratoxin A produced by *Aspergillus*,
- ochratoxin A, lemon and patulin produced by *Penicillium*.

These metabolites may be formed during the growing or harvesting season, and also as a result of improper storage of the crops. They are found in many agri-food products, such as: cereals and their products, nuts, spices, coffee, cocoa, tea, dried fruit, beer, wine, and milk (Solarska et al., 2012; Szulc et al., 2012). Economic losses resulting from mycotoxin contamination of grains, oilseeds, and feeds amount to nearly a billion dollars in the United States (Task Force Report 2003).

12.5. TRICHODERMA – ROLE IN PROTECTION AGAINST PHYTOPATHOGENS

When analyzing the results of studies on the presence of fungi in sewage sludge (Bień and Nowak, 2014), it should be stated that the sludge from municipal wastewater treatment can be a source of potentially phytopathogenic fungi. The most frequently isolated fungi are: *Acremonium*, *Altenaria*, *Aspergillus Cladosporium*, Fusarium and Mucor, which may pose a threat to the agricultural use of this waste. At the same time, it is noted that the types of strong antagonists of phytopathogens, such as *Trichoderma* (Brotman et al., 2010), often dominate among the fungi detected, the presence of which may reduce this risk. Figure 12.10

and 12.11 show the morphological structure of *Trichoderma harzianum* (Fig. 12.10) and the influence of *Trichoderma* viride on plant pathogens of the genus *Fusarium*, *Altenaria*, *Colletotricum* and *Pythium* in laboratory cultures (Fig. 12.11).

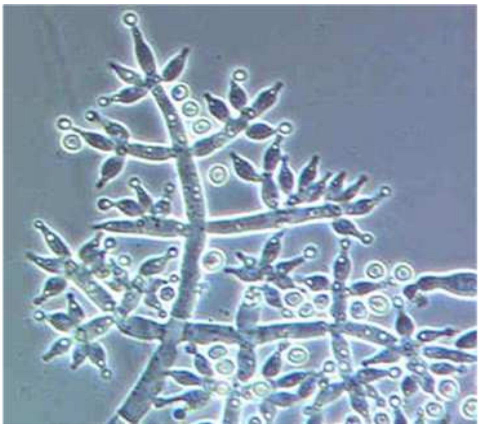

Fig. 12.10. *Trichoderma harzianum* – microscopic photography (https://alchetron.com/Trichoderma-harzianum)

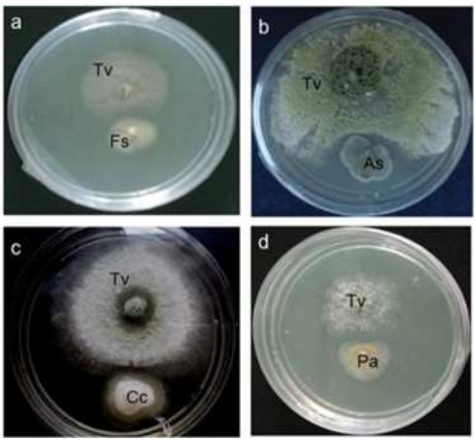

Fig. 12.11. Displays the Antagonistic effect of *Trichoderma viride* [Tv] against plant pathogenic fungi: a) *Fusarium solani* [Fs], b) *Alternaria solani* [As], c) *Colletotricum capsici* [Cc], d) *Pythium aphanidermatum* (Talapatra et.al., 2017)

Trichoderma virides is antagonistic against all plant pathogenic fungi used in the experiment. The inhibition zone was the largest in relation to *Colletotricum capsici*, and the smallest for *Altenaria solani*, after 14 days of incubation. In the soil environment, *Trichoderma* grows between the first two layers of plant root cells – the host. The proteins and enzymes produced by the fungus stimulate the production of signaling molecules that trigger a protective response in the plant when the pathogen is attacked (Fig. 12.12).

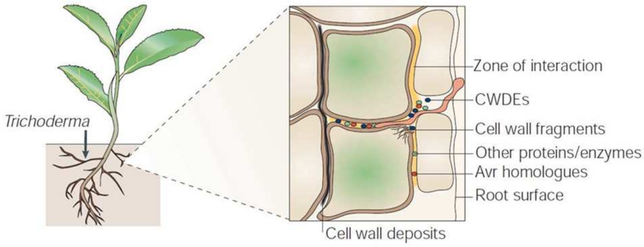

Fig. 12.12. Location of Trichoderma in the root of the plant (https://mycosolutions.ch/en/trichoderma-atrobrunneum/)

The antagonistic interaction between populations of different species is well known and a naturally occurring phenomenon in ecosystems. It is believed that fungi of the genus *Trichoderma* efficiently compete with plant pathogens for nutrients and compete effectively due to the high activity of hydrolytic enzymes in the root zone of plants (Hermosa et al., 2012). These fungi contribute to the stimulation of the growth and development of the root system and above-ground parts of plants, and consequently to their more efficient yield. Research (Błaszczyk et al., 2014) shows that the presence of the *Trichoderma* species in the rhizosphere and tissues leads to increased plant resistance to biotic and abiotic stresses. The natural antagonistic potential of *Trichoderma* species towards some phytopathogens and the presence of these microorganisms in the form of plant symbionts and endophytes result in these fungi being regarded as important factors in BCA (Biological Control Agent) biological control. They contribute to the reduction of the population of pathogens in the agricultural environment and the protection of plants and the improvement of their functional properties. Important abilities of Trichoderma fungi are mycoparasitism and antibiotics, as well as secretion of various lytic enzymes and secondary metabolites that inhibit the growth of phytopathogens (Vinale et al., 2014). Figure 12.13 shows the mechanisms of biocontrol exerted by the *Trichoderma harzianum* strain on *Guignardia citricarpia*.

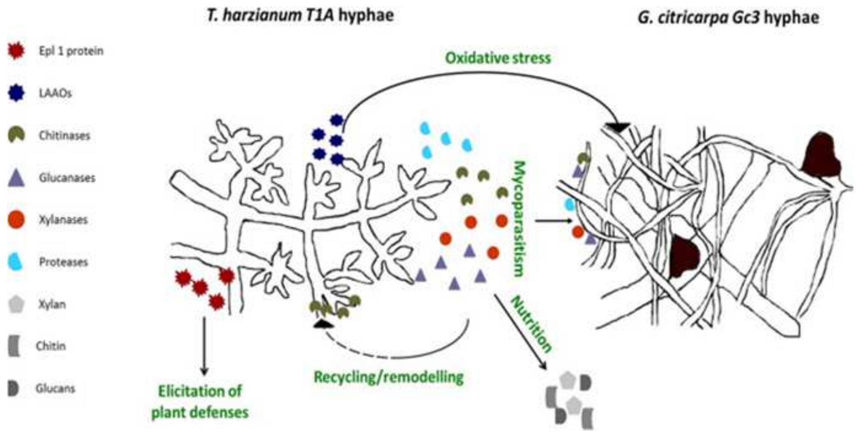

Fig. 12.13. *Trichoderma harzianum* T1A antagonism mechanism (Blauth de Lima et al., 2017)

The observed ability of fungi of the genus *Trichoderma* to mycoparasitism enabled the development of commercial biopreparations to combat phytopathogens (Table 12.1).

Table 12.1

Examples of preparations based on *Trichoderma* strains

Biopreparation	Strein used	Phytopathogen
Bio-fungus (Belgium) Trichoderma (Israel)	*Trichoderma harzianum*	*Sklerotinia, Fusarium* *Rhizoctinia, Pythium*
Trichodex	*Trichoderma harzianum*	*Botrytis, Sclerotinia* *Cladosporium, Sphaerotheca*
SoilGard (USA)	*Trichoderma harzianum*	*Fusarium, Rhizoctinia,* *Pythium, Sclerotinia*
BinabT (Sweden, USA)	*Trichoderma harzianum*	*Fusarium, Rhizoctinia,* *Pythium, Sclerotinia*
Trichopel, Trichoject Trichodowels, Trichoseal (New Zealand)	*Trichoderma harzianum* *Trichoderma viride*	*Fusarium, Rhizoctinia,* *Pythium, Armilaria*

Apart from the filamentous fungi of the genus *Trichoderma*, yeasts and certain species of bacteria also have inhibitory effects on phytopathogens (Liu J. et al., 2013; Sharma et al., 2009). In the case of yeasts, the mechanisms of biocontrol are associated with a high reproductive potential and the ability to quickly colonize the living space. This makes it possible to limit the development of pathogens on the plant surface. Yeasts limiting the development of phytopathogens belong to the genus *Aureobasidium pullulans* and *Pichia caribbica*, which inhibit the growth of pathogenic fungi belonging to *Botrytis cinerea*, *Penicillium expansum*, *Rhizopus stolonifer*, *Aspergillus niger* (Cao et al., 2013). The above facts indicate that sewage sludge, and in particular sewage sludge composts, are the source of the desired microorganisms and thus can be used in the biological control of phytopathogens. The sediments to be applied to the soil must meet certain conditions regarding the content of heavy metals and sanitary condition. These requirements are regulated by the Regulation of the Ministry of the Environment of February 6, 2015. Depending on the type of composted material, it is possible to inhibit the growth of such fungi as: *Fusarium oxysporum*, *Pythium ultimum*, *Verticillium dahlie*, *Pyricularia oryzae*, *Rizoctonia solani* (Cotxarrera et al., 2002). Positive effects were found in biocontrol of plant pathogens using composts prepared on the basis of sewage sludge with the addition of commercial bark or with the addition of vegetable, municipal and meat industry waste. This is also confirmed by the research conducted by Cwalin-Ambroziak et al. (2010), in which composts from sewage sludge and municipal waste were used for the cultivation of potatoes, wheat, barley and rape. The research was conducted for four years, analyzing changes in the abundance of phytopathogens belonging to the *Pythium*, *Phytophtora*, *Fusarium genera* and anatgonistic fungi of the *Trichoderma genus*.

The results clearly confirmed the increase in the number of *Trichoderma* in soils fertilized with composts from sewage sludge, with a simultaneous reduction of phytopathogens. The highest number of pathogens was present in the control soil without the addition of compost. Sludges from dairy wastewater treatment may be of particular use. They are characterized by a significant content of important antagonists of phytopathogens of the genus *Trichoderma* and a lower content of toxic chemicals.

12.6. CONCLUSION

In summary, the natural management of sewage sludge, i.e. the use of: plant cultivation, compost production and reclamation of degraded areas, gives the possibility of limiting the development of pathogenic microorganisms for plants. The biological activity of biowaste inhibits or completely prevents the development of harmful factors infecting plants, both during the growth and storage of crops. This method of phytopathogen control is friendly to the soil environment and allows for the reduction of chemical plant protection agents-fungicides. Fungicides, i.e. fungicidal plant protection products, are not indifferent to the natural environment, and their overuse disturbs the natural soil microflora. Long-term use of fungicides not only has a negative impact on the environment, but also, over time, leads to pathogens becoming resistant to these substances.

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Chapter 13

Collection of data on technologies for the production of sludge/biowaste products for agricultural applications

Agnieszka POPENDA

13.1. INTRODUCTION

The treatment and disposal of sewage sludge is a growing, environmental problem. It is commonly known that sludge production will increase as new sewage treatment plants are built. The challenge is to discover cheap and innovative solutions in the aspect of environmental quality standards (included in legislation acts) have become more stringent. The major outlets for sludge are agriculture and landfill, with only a relatively small amount being incinerated (Metcalf, 2003). Growing global population, urbanization and economic growth, together with production and consumption results in an increasing amount of solid waste all over the world as well. Solid waste management is regarded as a fundamental challenge of the twenty-first century (Lohri, 2017). Taking into consideration the aforementioned types of sludge, treatment and disposal options, application of biowaste and biosolids in agriculture are briefly described below.

13.2. TYPES OF SLUDGE, TREATMENT AND USES

The characteristics of sludge depend on the origin and quantity of flushing water (public or private toilet), its collection type (on-site, off-site) and level of treatment (Table 13.1). Excreta, called fecal sludge, gets collected in a toilet remain either on-site or is transported off-site in sewer systems. Sewage treatment plants produce sewage sludge, when suspended solids are removed from the wastewater and when soluble organic substances are converted to bacterial biomass which also become part of the sludge (Metcalf, 2003).

Raw sludge contains many pathogens, a high proportion of water, high biochemical oxygen demand (BOD) and is normally putrid and odorous. However, sludge also contains valuable nutrients (e.g. nitrogen and phosphorus), and it is a beneficial fertilizer. The organic carbon present in the sludge, after the process of

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stabilization, can be applied as a soil conditioner or can be transformed into energy through biodigestion or incineration.

Table 13.1

Properties of untreated and digested sewage sludge (Sagasta, 2015)

Item (dry weight)	Untreated primary sludge		Digested primary sludge	
	Range	Typical	Range	Typical
Total dry solids	2-8	5	6-12	10
Volatile solids	60-80	85	30-60	40
N	1.5-4.0	2.5	1.6-6.0	3.0
P_2O_5	0.8-2.8	1.6	1.5-4.0	2.5
K_2O	0-1	0.4	0-3	1
pH	5-8	6	6.5-7.5	7

The main purposes of sludge treatment are to reduce the water content and organic matter, BOD, pathogens and odors. Options for sludge treatment include thickening, dewatering/drying as well as stabilization/composting (Koné et al., 2009; Strauss et al., 2003). Water content in fresh sludge is as high as 98% which makes it unsuitable for composting. With sludge thickening, water content can be lowered up to 90%. Dewatering and drying reduce the water content further so that the solid part of the sludge remains about 20% (UNEP, 2001). Dewatering is faster but requires energy to press-filter or centrifuge while drying takes more time (even weeks) but does not require energy as water is lost through evaporation and drainage. Both aerobic and anaerobic processes can be used for sludge stabilization. Aerobic stabilization is typically done through composting at higher temperatures (55°C) which simulate an accelerated natural process that takes place on a forest floor where the organic material (leaf litter, animal wastes) is broken down, resulting in an overall reduction of volume, or converted to more stable organic materials. In anaerobic stabilization, bacterial decomposition through anaerobic processes, reduces BOD in organic wastes and produces a mixture of methane and carbon dioxide gas (biogas). Once properly treated, sewage sludge is called biosolids and, if safe, can be used e.g. in landscaping. Sludge can also be used for energy recovery, if sufficiently dry, directly through incineration or indirectly through anaerobic digestion, pyrolysis or gasification, which produce biofuels such as methane-rich biogas, biooil and syngas (Kalogo, 2012). Anaerobic digestion is the cheapest option as there is no energy input needed and the residual ‘cake’ can still be used as soil ameliorant. However, when sludge has high concentrations of heavy metals or persistent pollutants, anaerobic digestion would not be the best option as the resulting digested sludge would not be suitable for agricultural application. In these circumstances, incineration, pyrolysis or gasification may be more suitable.

It should be pointed out that sludge treatment should be applied to achieve the most and cost-effective technology. In Table 13.2, sludge treatment and disposal options are given. Some of the treatment options also achieve high removal of pathogens (e.g., thermal hydrolysis, lime addition, thermal drying and composting), and such processes may be increasingly necessary to secure sludge use on land.

Table 13.2

Sludge treatment and disposal options (Hall, 2020)

Sludge treatment		
Aims of treatment	Options	Examples
Conditioning	Chemical Thermal	Iron salt addition Lime addition Hydrolysis
Separation of phases	Thickening Mechanical dewatering Drying	Thickener Belt press Filter press Centrifuge Drying drum Dryer disc
Conversion	Biological Thermal	Anaerobic digestion Aerobic digestion Composting Pyrolysis/gasification Incineration Vitrification
Sludge outlets		
Integration in material cycle		Removal from material cycle
Use on land Resource recovery		Landfill Atmosphere (CO_2)

The options available for the use and disposal of sludge, and their practical benefits and constraints are included in Table 13.3.

In the aspect of agricultural application, there will be an increasing need to use advanced treatments to remove pathogens (e.g., *E. coli 0157*, *Salmonella spp*., etc.) (Hall, 2020). The agricultural outlet is vulnerable to adverse publicity and voluntary as farmers' requirements are seasonally variable. The aforementioned aspects make agriculture a precarious outlet and will be more difficult and costly by any further tightening of quality standards. Application of sludge to forests is not acceptable due to their special ecology, but in intensive timber and short rotation coppice wood production, sludge can be beneficial. In land reclamation, although here the need for large one-off applications to rapidly establish soil fertility should be recognised (Hall, 2020). Sludge intended for amenity and horticultural uses needs to be treated to a high standard of odour, pathogen and litter removal due to the likelihood of public access in treated areas. Incineration is a high cost/high technology option and is currently only likely to be cost-effective for large cities (Hall, 2020).

Table 13.3

The options for the sludge use land and fuel-based benefits and limitations (Hall, 2020)

Options	Benefits	Limitations
Sludge use options – land based		
Agriculture Reclamation Silviculture Forestry Amenity Horticulture	Policy Nutrients Organic matter Low cost/low technology	Voluntary Vulnerable Variable demand Quality Impacts Competition
Sludge use options – fuel based		
Incineration Supplementary fuel for power and processes Gasification	Green' energy Transport costs (if on site) Continuous process	Public perception Planning controls Costs Emissions Ash disposal

13.3. APPLICATION OF BIOWASTE IN AGRICULTURE

Biowaste is non-liquid waste with the high fraction of organic waste (usually more than 50% of the total waste) coming from households, restaurants, hotels, schools, hospitals, market waste, yard and park waste, and residues from food and wood processing industries (Hoornweg, 2012). Untreated biowaste causes hazard to public and environmental health by attracting insects, rodents and other disease vectors. Additionally, biowaste generates leachate that may pollute surface and groundwater supplies (Reddy, 2012). Furthermore, uncontrolled disposal of biowaste emits methane, a major greenhouse gas (Bogner, 2011). Biowaste treatment in a circular economy addresses resource scarcity, for instance the depleting of nutrients stocks such as phosphorus (Zabaleta, 2011).

Treatment technologies for urban solid biowaste can be divided into four groups: (1) direct use, (2) biological treatment, (3) physico-chemical treatment, and (4) thermochemical treatment (Lohri, 2017). Biowaste in agriculture can be processed as a source of carbon and plant nutrients into various soil amendments with benefit for both crops and soils. However, with increasingly intense agricultural practices, soils are progressively vulnerable, especially in tropics. Carbon turnover is 3-5 times faster than in temperate regions and extraction, decreasing nutrient retention and water storage capacity, and decreasing erosion resistance requiring replenishment of carbon and plant nutrient. Literature date on direct land application (land spreading) usually describe the spreads of raw agricultural waste (manure and/or crop residue) onto fields. Land spreading is relevant for crops requiring a large amount of organic nutrients (Dulac, 2001). Land application should focus only on pure organic waste as non-biodegradable waste fractions or pollutants would affect soil and crop quality or endanger farmers' health. According to a study carried out for the EU Commission, more than 90% of the waste spread on European land is agricultural waste and is mainly animal manure. The remaining 10% is food waste (Lohri, 2017). When considering urban organic waste, studies have shown the potential benefit of using yard waste and municipal organic waste (EPA, 2004) that can enhance organic matter levels, total nitrogen and available phosphorous in soils. The advantages and disadvantages of biowaste land spreading are given in Table 13.4.

Table 13.4

Advantages and disadvantages of biowaste land spreading (Lohri, 2017)

Advantages	Disadvantages
Raw organic waste undergoes natural aerobic biodegradation after it is spread onto the field. Degradation mobilizes nutrients and increases organic matter content of soil. Organic matter plays the following roles in soil: biologically acting as nutrient and energy supply for microbes, chemically buffering changes in soil pH capacity, as well as physically influencing soil structure and associated properties	Degradation may also cause a nitrogen competition in soil when the microbial population outcompetes the crop in the use of nitrogen for their own metabolism, as a result the crop shows signs of nitrogen deficiency
Soil amendment with high organic matter content	Biowaste may result in leaching of nutrients into groundwater or surface water or the volatilization as ammonia
Land spreading is beneficial for degraded soils in arid areas	Direct land application of waste is not a treatment process and might negatively impact on plants and soil. Waste is likely to contain a certain level of pathogens or trace elements, these can bio-accumulate in plants and soil. This may result in health threats from food contamination or pollution of water courses from runoff
	Land spreading causes the risk of lower availability of micro-nutrients necessary for plant growth when applying non-stable organic material
Land spreading of raw organic waste requires control to avoid environmental and human health risks (Dulac, 2001; EPA, 2004). One control measure is to ensure sufficient time between application of waste and the subsequent crop planting and harvesting	

Land spreading of raw organic waste can be applied in rural areas of low-and middle-income countries for improving soil nutrients content. The advantages and limitations of this practice depend on the quality of the waste. As land spreading does not remove pathogens, spreading of plant disease to plants and farming workforce related health is threatened. If the waste is polluted with e.g. heavy metals, these may be present in soils or crops. The investigations on direct land application take into account the impact of organic residues on soil and/or crop structure and trace element content (Walsh, 2012). Organic residue properties are highly variable as are the soil and crop response. Alvarenga et al. (2007) claim that eco-toxicity tests combined with chemical analysis allow for a good environmental risk assessment of direct land application for evaluating contaminant bioavailability, mobility and toxicity. The important issue is that the time period between land spreading and planting of crops should be sufficiently long to ensure minimal risk for soil and plants. In urban areas, this issue can be difficult to follow. In general, it is recommended to avoid direct land application but rather include a treatment process (e.g. through composting) before spreading the waste onto the field. This ensures a hygienization phase and the conversion of nutrients into a more readily available form for the plants. In Table 13.5, the input and output of land application is given.

Table 13.5

Direct land application input and output (Lohri, 2017)

Input		**Output**	
Feedstock	Biowaste	Value products	Soil amendment
Conversion Operating conditions	Not specified	Product yield	Not specified
Resource requirements (water, energy, space, etc.)	Large land requirement	Comments	Salt, heavy metals, chemical components may affect crop
Processing time	Not specified	Prevalence in low and middle income settings	Widespread practice
Hygienization	No hygienization		
Emissions	CO_2 and water vapor		
Skill requirement	Only simple labor skills required		

13.4. BIOSOLIDS FOR AGRICULTURAL APPLICATIONS

Biosolids are organic additives produced during wastewater treatment that can be used as soil amendments in order to supply nutrients for plant growth including nitrogen and phosphorous, as well as some essential micronutrients such as nickel, zinc, and copper (Metcalf, 2003). The nutrients present in the biosolids are regarded as being valuable as they are organic and released slowly to growing plants. Due to the low solubility in the water the organic forms of nutrients are less hazardous to groundwater or run off into surface waters. The beneficial use of biosolids also includes support of soil with organic matter. Biosolids can be applied on agricultural land, forests, or on reclamation of land. Biosolids can also be treated as option for expensive chemical fertilizers. The methods of applying of biosolids depend on the type of land and water contents. Liquid biosolids contain 94 to 97% water and low amounts of solids (3 to 6%). These can be injected into the soil or applied to the land surface. Through mechanical processes such as: draining, pressing, or centrifuging, the amount of water in biosolids can be reduced of up to 30% dry solids. According to EPA requirements, wastewater solids have to be treated before agricultural application. The aim to process of "stabilization" is to minimize generation of odour, destroy pathogens and reduce vector attraction potential. The methods of wastewater solids stabilization include: digestion, composting, heat drying, adjustment of pH or alkaline stabilization (Metcalf, 2003). The advantages and disadvantages of land application of biosolids are included in Table 13.6.

Table 13.6

Benefits and limitations of land application of biosolids (Alvarenga, 2007; EPA, 2000; Metcalf, 2003)

Benefits	Limitations
Recycle wastewater solids as long as the material is quality controlled	Limited especially in colder climates. Biosolids should not be applied to frozen or snow covered grounds
Returns nutrients to the soil and enhances conditions for vegetative growth	Spring rains can make impossible to get application equipment into farm fields
Relatively inexpensive compare to other technologies	Potential public opposition (if it is close to residential areas). Odor is the problem
Spatial needs can be relatively minor depending on the chosen method of stabilization	Municipalities or counties may pass ordinances which ban or restrict the use of biosolids

Land application can have both positive and negative impacts on water, soil, and air (Metcalf, 2003). Environmental impacts of biosolids are given in Table 13.7.

Site suitability depends on: soil characteristics, slope, depth to groundwater, and proximity to surface water. Additional requirements to protect water quality include: sufficient land to provide areas of buffers around surface water bodies, wells, and wetlands, depth from the soil surface to groundwater equal to at least one meter, and the pH of soil in the range of 5.5 to 7.5 to minimize metal leaching and maximize crop growing conditions. The type of vegetation results in the choice of application equipment, the amount of biosolids to be applied, and the timing of applications. Time of biosolids application is also very important and must not interfere with the planting of crops. Application is most beneficial on agricultural land in late fall or early spring before the crop is planted. Timing is less critical in forest applications when nutrients can be incorporated into the soil throughout the growing period. Winter application is less desirable in many locales. Applications of biosolid can be made as long as the ground is not saturated or snow covered and whenever livestock can be grazed on alternate lands for at least 30 days after the application.

Table 13.7

Environmental impacts of biosolids (O'Dette,1996)

Advantages	Disadvantages
The potential for nitrogen compounds to leach from biosolids amended soil is less than that posed by the use of chemical fertilizers by proper management practices (the application of biosolids at agronomic rates)	Excess of nitrogen compounds in the biosolids can leach from the soil and reach groundwater
Biosolids stabilization reduces odors and usually results in an operation that is less offensive than manure application	Runoff from rainfall may carry excess nutrients to surface water
A properly managed biosolids land application program is preferable instead of using conventional fertilizers	Odors from biosolids applications are the primary negative impact to the air Stabilization processes such as digestion can decrease the potential for odor generation
Biosolids are a recycled product, use of which does not deplete non-renewable resources such as phosphorous	
The nutrients in biosolids are not as soluble as those in chemical fertilizers and are therefore released more slowly	
Often subject to more stringent soil conservation and erosion control practices, nutrient management, and record keeping and reporting requirements than farmers who use only chemical fertilizers or manures	
Closely monitored	
The organic matter improves soil properties for optimum plant growth	
Decreasing the need of applying pesticide	

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Chapter 14

Evaluation of the effectiveness and stability of assessment indicators and modelling of the degree of organic carbon sequestration of soil

Krzysztof REĆKO

14.1. INTRODUCTION

At present, human activity is an important factor causing global warming. The carbon cycle is unbalanced and, as a consequence, there are challenges like global warming and the greenhouse effect occur. Rising level of carbon dioxide (CO_2) in the atmosphere changes the climate by heating the Earth and increasing the frequency of extreme weather events. The amount of carbon dioxide in the atmosphere increases due to the burning of fossil fuels, deforestation and changes in soil use (Cowie, 2009; Dhanwantri et. al., 2014).

It is believed that the increase in soil temperature due to warming will accelerate the mineralization of nutrients from organic matter contained in the soil, which will increase plant growth. The total carbon content of the ecosystem will depend on the balance between plant growth and their degradation rate. Due to the fact that soils currently contain about 1500 Gt C, and about 1/3 of it is stored in arctic and boreal soils, there is a concern that warming these soils may release a significant amount of carbon into the atmosphere, increasing its inflow as a result of burning fossil fuels and changes in method use of soil and accelerate the warming process. Still, it is believed that soil organic carbon sequestration can have many benefits if appropriate practices are in place to promote this process. Soil organic carbon (SOC) modelling is therefore one of the most important tools for determining the impact of organic material management for its sequestration. Using soil organic carbon models, it is possible to assess the impact of future climate change on soil SOC and predict its stocks. Using soil organic carbon models, it is possible to assess the impact of future climate change on changes in soil levels and to predict its reserves. Due to the complexity ofsoil organic carbon modelling systems, simulation models can be useful for studying soil-plant-atmosphere relationships and improving the transparency of SOC dynamics (Cowie, 2009; Lal, 2011).

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14.2. SOIL ORGANIC CARBON

Soil organic carbon is a measurable component of soil organic matter and plays an important role in all soil processes. Soil organic carbon levels are directly related to the amount of organic matter contained in soil. Soil organic matter (SOM) contains plant and animal residues at various levels of mineralization and humification, as well as organic products of living soil organisms. The source of carbon (C) is the above-ground and underground plant biomass introduced into the soil, as well as organic mass in the form of manure, slurry, composts and green manure (Gonet, 2007; Sapek, 2009). Organic carbon in the soil is in the form of humic substances, which are resistant to degradation, from which humus forms as a result of humification. Organic carbon compounds and their combinations with minerals may occur in form of insoluble (C-org) and soluble organic carbon in soil, therefore land use may result in its preservation or loss (Pikuła, 2019).

Soil organic carbon levels are affected by processes such as photosynthesis, respiration and decomposition. Other processes that can lead to carbon loss include soil erosion and leaching of dissolved carbon into groundwater. When carbon inputs and emissions are in balance, there is no net change in soil organic carbon levels. When carbon inputs from photosynthesis exceed C losses, soil organic carbon levels increase over time (Ontl and Schulte, 2012).

The importance of organic carbon in soilsshould be understood more broadly than merely production functions. Organic carbon included in soil organic matter (OM) shapes important soil functions such as nutrient changes, structure and aggregation, permeability, sorption and filtration capacity, buffering, activity of soil which affect their productivity (Gruszczyński, 2014; Merante et al., 2014).

Soils are moving towards the "equilibrium" level, determined by the balance between the amount of C-org that reaches the soil with plant residues, and the loss of C-org from the soil, primarily by decomposing organic. Changing the rate of C-org inflow to the soil and / or the rate of decomposition causes an increase or decrease in soil C reserves, and consequently a change in the value of the soil equilibrium level (Wójcik, 2013).

The main factors threatening the sustainability of the organic carbon pool in soil are land use change or unsustainable agricultural activities, reckless and unnecessary deforestation, drainage of organic soils, soil cover removal, replacing natural ecosystems with agricultural systems favoring intensification of water and wind erosion (Gruszczyński, 2014; Van-Camp et al., 2004).

Complementing these losses is possible in most soils between 60-80%. However, increasing the organic carbon content (C-org) is more often possible in newly developed ecosystems than in natural ecosystems, due to the limitations resulting from the properties of their soils (Lal, 2001; Lal, 2004).

Soil organic carbon losses also occur as a result of leaching of soluble organic carbon to water as a result of water erosion. The soluble form of carbon is part of one of the most mobile and fastest decomposing soil fractions of organic matter, i.e. soluble organic matter and plays a significant role in the environment, because it increases the loss of carbon from soil. The soluble fractions of organic matter may be washed away and then adversely affect the quality of surface and groundwater (Pikuła, 2019).

The greatest amount of organic carbon can accumulated the forest use of soils, i.e. the conversion of arable land to non-agricultural land. Significant organic carbon reserves also maintain permanent grassland, especially on organic soils. Resources accumulated in arable land are slightly lower due to cultivation operations and a long period of lack of soil cover, limiting the process of carbon loss (Gruszczyński, 2014; Pikuła, 2019).

The resources of organic carbon in intensively used agricultural soils constituting large areas of the global land surface tend to decrease. A negative carbon balance in some regions of the EU contributes to the deterioration of soil functioning, but also to an increase in CO_2 emissions, worsening the balance of greenhouse gas emissions from the use of agricultural land. Despite this, agriculture is considered to have great potential for organic carbon sequestration in soils if improved management practices conducive to this process are pursued (Banwart et al., 2015; Faber and Jarosz, 2018).

14.3. SOIL ORGANIC CARBON SEQUESTRATION

The Intergovernmental Panel on Climate Change (IPCC) reports suggest that even if significant reductions in anthropogenic CO_2 emissions are achieved in the near future, carbon sequestration will be needed to ensure a safe level of CO_2 in the atmosphere and to mitigate climate change (Kane, 2015; Lefèvre et al., 2017).

Depending on processes and technological innovations, there are three main types of C sequestration (Lal, 2011):

- based on the natural process of photosynthesis and the transformation of atmospheric CO_2 into biomass, soil organic matter or humus and other components of the terrestrial biosphere;
- engineering techniques for carbon capture and storage;
- chemical transformations.

Carbon sequestration in terrestrial and aquatic ecosystems based on natural processes is more profitable and brings many additional benefits compared to engineering techniques and the conversion of CO_2 to carbonates. Soil organic carbon sequestration can contribute to maintaining food security and at the same time will have a significant impact on the concentration of CO_2 in the atmosphere, and therefore will affect climate parameters. In addition, it helps maintain soil structure and improves its quality, reduces nutrient loss, retains water and reduces soil erosion (Banwart et al., 2015; Lal, 2011).

Sequestration (soil organic carbon binding) is a process in which carbon is bound to the atmosphere by plants or organic residues and accumulated in soil. Therefore, soils play a huge role in maintaining the balance of the global carbon cycle. Maintaining the ability of soil to sequester carbon is one of the most important tasks of EU environmental policy (Lefèvre et al., 2017; Sapek, 2009).

The size of the global soil organic carbon pool is spatially and temporally variable and determined by abiotic and biotic factors. Estimates of global soil organic carbon resources vary depending on latitude, climate, land use and management. Most of the global SOC is stored in northern latitudes, especially in permafrost and humid boreal areas (Kane, 2015; Zomer et al., 2017).

The global distribution of soil organic carbon is strongly influenced by temperature and precipitation. It is generally lower in the tropics, where it is warmer and / or drier, and higher in colder, wetter areas, more northern and slightly less in southern latitudes. Warm, dry conditions favor the accumulation of SOC due to the high efficiency of biological processes, while cool, humid conditions favor the accumulation of SOC due to much slower decomposition. The amount of organic carbon in soil varies significantly depending on geographical location and land cover. Peat bogs have exceptionally high SOC content but cover less than 0.3% of the global earth's surface. On the other hand, Savannah has relatively low SOC content, but covers a large area around the world (Banwart et al., 2015; Lefèvre et al., 2017).

14.4. INDICATOR OF CARBON SEQUESTRATION

Soil organic carbon content (the amount of carbon stored in soil) is commonly used in indicators representing soil organic matter and together with it affects soil quality, its functions, and fertility. Most of SOM is in the topsoil layer. The concentration of SOC in surface soil is necessary to control erosion, infiltration of water and nutrients, which can easily be affected by changes occurring in various ecosystem processes. The concentration of SOC in deeper soil layers is relatively stable and is usually used as a reference point when making comparisons of soil from different ecoregions or research sites with inherent differences in soil capabilities. Dry and semi-dry climatic conditions, where crop residue management is of paramount importance for achieving sustainable plant production, can be a factor limiting the accumulation of organic carbon in the upper soil layers. SOC concentrations may be associated with changes in SOC sequestration, and therefore the stratification index (SR) of organic carbon in soil. It is defined as the ratio of the concentration of SOC in the surface layer of the soil to the concentration of SOC in the deeper soil layers well below the surface. Has been proposed as the sequestration indicator SOC and soil quality changes in various natural ecosystems and management practices soil (Sá and Lal, 2009; Six et al., 2000).

Soil disturbances resulting from soil cultivation are the main reason for the depletion of organic matter and the reduction in the number and stability of soil aggregates when native ecosystems are transformed into agriculture. Non-tillage and conservation tillage systems usually show increased aggregation and soil organic matter over conventional tillage. The soils in these systems remain undisturbed, the soil aggregates remain intact, physically protecting carbon (Kane, 2015; Six et al., 2000).

João Carlos de Moraes Sá and Rattan Lal (2009) conducted a study in which they assessed changes in soil organic carbon stratification over time, taking into account different cultivation method. The research material consisted of soils classified as dark red loam, having a deep and very well-structured structure, high porosity and very good internal drainage, which is characterized by low natural fertility. They were used chronologically, i.e. first as native fields with natural vegetation, then transformed into arable fields by plowing and cultivating traditionally using a rotational crop cycle. They analyzed soil samples for chemical and mineralogical substances. Soil samples were taken from four soil depth levels (0-5, 5-10, 10-20 and 20-40 cm). The test results showed that the concentration of organic carbon in the soil, nitrogen total (TN) and sulfur total (TS) concentrations decreased with soil depth, and they had a different distribution in the soil profile over the chronological course of the cultivation. The continuous tillage process thus leads to a stratification of the SOM pool with the highest accumulation in the surface layer (Sá and Lal, 2009; Singh, 2019).

Sá and Lal (2009) show that with an undisturbed course of soil cultivation, a natural process of accumulation of SOC, TN and TS in the surface layer of the soil occurred, and the stratification index increased with increasing soil depth. This was due to the introduction of litter to the soil surface, thus improving its state of elasticity. SR and SOC differed significantly between the depths, while the concentrations of N and S differed only when comparing the surface layer of the soil with its smaller depths (Franzluebbers, 2002; Sá and Lal, 2009).

According to Sá and Lal (2009), soil management by introducing appropriate native field cultivation practices affects SR regardless of climate and soil type. Taking SR of 2.5 as the reference for soil quality, they observed a 150% difference between the stratification index SOC for a depth of 0-5 : 5-10 cm, and the stratification index SOC for a depth of 0-5 : 20-40 cm. The increase in SR with increasing depth was observed by comparing the SR for the surface layer of the soil and its deeper layers in the case of different soil cultivation methods. The comparison was made taking into account fields with long-term zero-tillage, semi-native fields with natural vegetation, fields cultivated with plowing and those cultivated traditionally for a long time using a rotation cycle of crops. The obtained results indicated an increase in SR with increasing depth for long-term treatment without plowing and it was more than for native fields with natural vegetation.

Sá and Lal (2009) show that the total carbon input of 7.31 kg C m^2 over 22 years continuously supplied to fields with no tillage restored the SOC concentration above the level observed for semi-native with natural vegetation and increased SR (Sá and Lal, 2009). The use of long-term NT with continuous use as post-harvest litter residue leads to the accumulation of a thick surface layer indicating the importance of SR as a good index of sequestration C (Sá and Lal, 2009).

Soil cultivation causing loosening of the surface exacerbates soil susceptibility to erosion and thereby increases C losses by breaking aggregates and extending the time required to restore them (Franzluebbers, 2002). Sá and Lal (2009) show that the decrease in organic carbon and nitrogen particles reduces stable carbon and stable nitrogen, accentuating C emissions to the atmosphere by decomposition and reduced storage of C. According to them, in undisturbed soil, SR increased with increasing soil depth. The mean SR for organic carbon particles and nitrogen particulates was greater than for stable carbon and stable nitrogen. The soil without cultivation all year round, protected by the mulch, created ideal conditions for a continuous stream of C and N into the soil and sequestration of C (Sá and Lal, 2009).

Continuous introduction of crop residues onto the soil surface has provided environmental benefits, i.e. increased biological activity and stimulation of macroaggregation. SOC sequestration rates for the 0-10 cm layer indicated the importance of SR as a tool to estimate the speed and amount of SOC sequestration (Sá and Lal, 2009).

14.5. SOIL ORGANIC CARBON MODELLING

The complexity of the soil-plant-atmosphere system makes simulation models useful for studying system dependencies, determining the impact of organic materials management on SOC resources, predicting changes in SOC resources, and studying the impact of different management scenarios on SOC sequestration. Modeling can also be an important tool for estimating SOC gains from new soil management (Afzali et al., 2019; Gruszczyński, 2014).

The Intergovernmental Panel on Climate Change (IPCC) proposed a method with three levels of detail to take account of changes in resources and SOC dynamics caused by land use and land use change. They represent different approaches to methodological modelling, from the use of default data and empirical equations to the use of more complex, specific, and locally validated functional or mechanistic models (Jarosz and Faber, 2017; FAO, 2019).

IPCC inventory over a 20-year period in which the SOC inventory reaches a new steady state (called "balance"). This approach estimates the change in SOC stocks depending on climate, soil type and other factors. Default information on climate, soil use, management policy provided by the IPCC or, if available, country-specific data is used (FAO, 2019; Ponce-Hernandez et al., 2004).

Numerical models, including mathematical models, describe SOC changes and biogeochemical soil processes. However, due to their structure, the number of input variables required, and the temporal and spatial resolution, not all available C models are suitable for all studies.

The assessment of SOC changes on a global and regional scale is possible due to the use of models such as RothC, DNDC and CENTURY based on biogeochemical soil processes formulated in accordance with mathematical and ecological theory. These models are able to simulate SOC turnover according to specific soil conditions and link it to management practices. They refer to user-defined time and spatial scales based on scenarios characterizing the internal SOC dynamics (Ponce-Hernandez et al., 2004).

Model Rothamsted Carbon – Model RothC – is one of the models widely used to interpret fluctuations in SOC content in various climatic, soil and different usage conditions (Gruszczyński, 2014).

The RothC model has been a widely used model in many countries for many years and is a soil organic matter distribution modelused to simulate the dynamics of organic carbon in agricultural soils in various environments and management practices. Its key advantage, compared to other process-based models, is the need to provide only basic input data that is easily accessible. The set of input data required by the RothC model are mainly: climatic data (i.e. monthly rainfall, monthly evapotranspiration, average monthly air temperature), soil data (i.e. clay content, initial soil organic carbon, stock, considered depth of soil layer) or land use and land management data (i.e. soil cover, monthly plant residue input, monthly manure input, plant residue quality factor). However, there are examples of RothC applications on a continental or global scale, with either low spatial resolution or a low number of LU classes (Afzali et al., 2019; Barančíková et al., 2010).

The RothC model has been widely and often used, in all cultivation systems in the world, to simulate changes in SOC resources in soils under various management methods and climatic conditions. In addition, it was used on measurements obtained from 16 long-term experiments throughout the entire crop area and showed good overall results in representing SOC dynamics (FAO, 2019).

Using the RothC model and the latest state of current soil and climate databases, Guocheng Wang et al. (2017) presented a spatio-temporal simulation of SOC dynamics in the soils of major global cereal cultivation systems, i.e. wheat, corn and rice. Selected wheat, corn, and rice crops covered about 72% of global cereal crop areas and accounted for about 80% of global cereal crop production in the world. Carbon turnover in global arable soils was simulated in various input scenarios (monthly calculations from crop residues, roots and manure) in the time interval from 1961 to 2014 in high spatial resolution (Singh, 2019; Wang et al., 2017).

According to Wang et al. (2017), out of three scenarios, a relatively higher increase in SOC occurred in the central latitudes of the northern hemisphere (central parts of the USA, western Europe, northern regions of China). A relatively small increase in SOC occurred in regions of high latitude in the northern and southern hemispheres, while SOC decreased in equatorial zones of Asia, Africa and America. On a global scale, 69-89% of the study area acted as a net carbon sink in various residue management scenarios (Wang et al., 2017; Singh, 2019).

SOC dynamics simulation results suggested a generally linear relationship between carbon input and SOC variability. Although the soil can accumulate a significant amount of SOC when the existing carbon content in the soil is low, as soon as it reaches the state of saturation it ceases to be an absorber and further changes cannot occur even after adding more carbon. Natural soil features, i.e. the content and type of clay, have a strong effect on the saturation limit of the soil. Without taking saturation into account, the first order decay model (e.g. RothC) can cause a significant deviation in the simulation, especially for longer SOC time simulations in regions where the carbon input is higher and the SOC distribution is lower (Wang et al., 2017; Kane, 2015).

The CENTURY model is wider than the RothC model and can simulate a wide range of crop rotations and cultivation practices to assess the impact of management and land use change practices on productivity and sustainable development of agricultural ecosystems. It is used to present simulations of long-term SOC and nutrient dynamics, i.e. nitrogen (N), phosphorus (P) and sulfur (S) for various plant-soil systems (arable land, grassland, forests, savanna) (Ponce-Hernandez et al., 2004; Shrestha et al., 2009).

The CENTURY model operates using a monthly time interval, and input variables are available for most natural and agricultural ecosystems and can generally be estimated from existing literature. The main input variables for the CENTURY model are: average monthly maximum and minimum air temperature, monthly precipitation, lignin content in plant material, N, P and S content, soil texture, N inputs and initial levels of C, N, P and S in soil (Metherell et al., 2014; Ponce-Hernandez et al., 2004).

Using the CENTURY model by Shrestha et al. (2009) they assessed and analyzed the anticipated changes in the SOC pool in cultivated and managed forest land in the inland catchment area in Nepal for a period of 100 years (1950-2050).The catchment area was divided into two categories of land use: forest (59%) and arable land (41%). The arable land was rainy highlands (Bari) and lowland arable lands (Khet). The CENTURY model was used to simulate three types of utilized land: the rainy upland (Bari), irrigated lowland agricultural land (Khet) and dense forests for less than 100 years (1950-2050).

Shrestha et al. (2009) indicate that in the forest area, the SOC pool dropped sharply to a low level in 1972. After applying an improved management strategy, the decrease in the SOC pool was significantly slowed and began to remain almost constant until 2050 (Shrestha et al., 2009).

According to Shresth et al. (2009) using the scenario of a different cropping system, it was also observed that the early growing period without the addition of fertilizer in combination with high residue removal rates led to a sharp decrease in SOC. Then, depending on the scenario, after a period of an almost constant level, SOC increases began in the 1960s and 70s, which was influenced by the effect of increasing the amount of fertilizer.

Changing climatic factors, such as rainfall, temperature, and erosion, affect plant productivity, degradation rate, and thus a change in the SOC pool (Shrestha et al., 2009).

The model did quite well in representing the effects of different management systems in different areas of the study. Simulation results according to Shresth et al. (2009) showed significant loss of SOC from the system in the first-time block in all three types of land use included in this study. However, this has been maintained or improved at a later stage as part of better governance. The potential for partial renewal of SOC resources on arable soils was mainly due to the addition of manure and crop management systems. Soil C pools estimated by the

CENTURY model for various soil applications are comparable with the measured values. Only slight differences between the estimated and measured SOC pool were shown (Shrestha et al., 2009; Singh, 2019).

Process models simulating the dynamics of soil organic matterare recognized as valuable tools for the quantification and understanding of SOC dynamics in response to agricultural practices, particularly in the context of rapid policy changes and / or the extension of experimental research results at regional or national level (Dimassia et al., 2018; Shrestha et al., 2009).

14.6. CONCLUSION

Soil organic carbon sequestration is a way to mitigate climate change by reducing CO_2 emissions. SOC sequestration reduces carbon emissions and losses, improves soil and water quality, and reduces nutrient losses and soil erosion. SOC sequestration is possible due to many soil management strategies. To predict soil organic carbon levels, various SOC models are used as part of various soil management strategies.

SOC models are therefore essential for understanding soil processes, identifying needs and identifying the technical potential for the potential available for SOC sequestration, developing a framework for different management scenarios to optimize a pool of SOC networks or to identify multi-functional land use or soil management systems where sequestration carbon dioxide is an integral component.

Soil organic carbon sequestration indicators can be used as indicators of soil quality in various natural ecosystems and management practices. They provide information on how different farming practices and systems used on different soil types affect the soil's ability to absorb CO_2. They enable both estimation of actual SOC storage and forecast of SOC storage potential, which is an important aspect in land use planning and management.

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Chapter 15

Assessment of the effectiveness of bioremediation of degraded areas with the use of stabilized sewage sludge and other biodegradable waste

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15.1. INTRODUCTION

We are observing the effects of intensive activity of heavy industry and a progressing population growth in the world, especially in the last century, in the form of accumulation of pollution in the environment and the increasing amount of generated waste. Despite the growing public awareness of broadly understood environmental protection, a rational strategy for the reclamation of degraded areas and appropriate waste management is still a big challenge. Researchers have been increasingly focusing great attention to the pro-ecological approach to this issue, due to the growing importance of the so-called circular economy and the benefits of using biological methods. One of the main issues in the coming years will be the protection of natural resources through the integration of various systems: electricity, heating, cooling, transport, water, buildings, industry, forestry and agriculture. The closed loop of waste is based on the use of by-products from one technology as a raw material in another and precisely when it is beneficial for everyone. In this context, it is important to valorize and characterize the impact of waste on soil fertility and crop stimulation (Mosquera-Losada et al., 2019; Urbaniec et al., 2016).

Soil is a special element among terrestrial ecosystems because it enables the growth and development of plants and is also the environment of many microorganisms (Sas-Nowosielska, 2009). Since the industrial revolution, which began at the turn of the 18th and 19th centuries, there has been a significant reduction in the resources of this ecosystem as a result of degradation or severe contamination. These pollutants emitted mainly by industry and mining cause many negative occurrences in the soil environment with long-lasting and often irreversible effects (Fijałkowski and Kacprzak, 2009). A very important group of

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harmful substances is the so-called persistent organic pollutants, including organochlorine pesticides, polychlorinated biphenyls, polychlorinated dibenzo-dioxins, polychlorinated dibenzofurans, and polycyclic aromatic hydrocarbons. Most of these compounds can accumulate in the fat tissue of living organisms and cause serious health effects for humans due to their strong toxic, carcinogenic and mutagenic properties. Organic matter is a special element of soil because it affects the transformation processes of persistent organic pollutants. These compounds dissolve poorly in water, reacting with hydrophobic elements of organic matter, which reduces their mobility and bioavailability (Ukalska-Jaruga et al., 2015). Similar relationship occurs in the case of inorganic impurities, which include heavy metals. Areas contaminated with trace elements are very often characterized by low or no organic matter and adequate microflora. Increasing the share of organic fraction and restoring biological activity can contribute to significant improvement in soil quality and reduce heavy metal migration (Fijałkowski and Kacprzak, 2009). Given the importance of organic matter for the soil environment, the use of waste such as sewage sludge or other biowaste as an additive to land can be a beneficial move to remediate contaminated areas.

15.2. CAUSES OF SOIL ENVIRONMENT DEGRADATION

The majority of environmental pollutants accumulate in the soil. This phenomenon is alarming due to the special properties of this ecosystem, which is the source and storage of nutrients and water for many organisms. In addition, soil is a place of decomposition of organic matter ensuring the circulation of micro- and macroelements in nature. Toxic substances get into the soil along with precipitation, surface water discharges, and dust, as well as a result of industry, communication means and improper waste and agrochemical management (Stepnowski et al., 2010). Soil pollution can have its source in both natural processes, such as weathering of the parent rock, however anthropogenic activity is the main factor responsible for environmental degradation (Hamid et al., 2020). The accumulation of xenobiotics in the soil matrix poses a threat not only to the biodiversity of organisms inhabiting it, but also to human and animal health, due to the possibility of migration of these compounds (Kowalska and Łukaszyk, 2020; Rigoletto et al., 2020; Włóka et al., 2018). Impurities deposited in the soil can be sorbed, microbiologically decomposed, can be accumulated by plants, washed away into the soil profile of the soil, or escape with water vapor. These processes depend on the chemical structure of impurities and on the sorption properties of soils (Stepnowski et al., 2010). Figure 15.1 shows directions of migration of pollutants in the environment (Stepnowski et al., 2010).

Among the pollutants of soils, both organic and inorganic compounds can be mentioned. Organic xenobiotics that pose the greatest threat and damage to the environment include pesticides, petroleum substances, including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), dioxins and drug derivatives (Krosowiak et al., 2008; Włóka et al., 2018; Wyrzykowska et al., 2006). Among the inorganic substances that degrade areas, heavy metals, mineral nitrogen and aluminum are most often mentioned (Stepnowski et al., 2010). Table 15.1 presents the list of pollutants whose concentrations most often exceed the limit values in soils and the sources of their origin (Stepnowski et al., 2010).

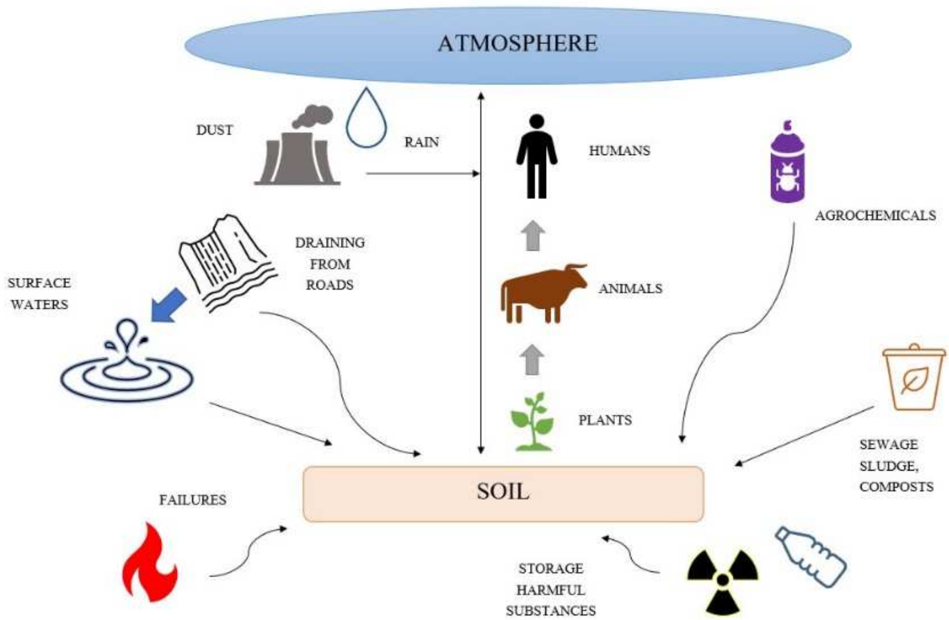

Fig. 15.1. Directions of migration of pollutants in the environment (Stepnowski et al., 2010)

In Poland, the majority of research on soil pollution is focused on the presence of heavy metals and petroleum compounds in soils. The toxicity of trace elements largely depends on the chemical forms in which they occur in the environment. Heavy metals occur in various physicochemical connections in soil, in the form of free ions or in mineral and organic compounds depending on the conditions in the environment. This translates directly into their mobility, i.e. the ability of the element or one of its chemical forms to move. The most dangerous for ecosystems are metals in a mobile form because they become more available to living organisms and easily migrate in the environment. On the other hand, the elements in stable form, i.e. those related to the crystal structure of soil minerals, pose the smallest threat (Kacprzak, 2013). In turn, PAHs originating from anthropogenic sources do not occur in the environment in the form of single compounds, as they always form multi-component mixtures. The quantitative and qualitative composition of these mixtures depends on the type of combusted material and the conditions in which the process of crude oil combustion and processing takes place (Stepnowski et al., 2010).

Table 15.1

List of pollutants whose concentrations do not meet soil quality standards and their sources (Stepnowski et al., 2010)

Source of pollution	Type of pollution
Oil refineries, gasification plants and coal liquefaction, oil wells, coking plants	Aliphatic hydrocarbons, PAHs, phenols, cresol
Ports and port installations for handling liquid fuels, chemicals, metal ores	Aliphatic hydrocarbons, gasoline, PAHs, pesticides and other organic pollutants, heavy metals
Chemical coal treatment plants	Aliphatic hydrocarbons, PAHs, phenols, cresols, cyanides
Plastic plants (glues, resins and polymers)	Phthalates, phenols, cyclohexane, hydrocarbons Chlorinated
Paint and solvent production plants varnishes	Aromatic hydrocarbons, hydrocarbons chlorinated, zinc, lead, chromium, barium
Power stations and switchboards	Polychlorinated biphenyls
Conventional power plants, combined heat and power plants, combustion installations	Polycyclic aromatic hydrocarbons, heavy metals
Iron foundries, steel and non-ferrous metals	Heavy metals, cyanides, phenols, hydrocarbons Aliphatic, PAHs
Ceramic plants	Cadmium, lead
Plants producing lighting articles and measuring	Mercury
Electroplating plants	Heavy metals, cyanides
Waste incineration plants and other decommissioning plants hazardous waste	Pesticides, aliphatic hydrocarbons and aromatic, PAH, heavy metals
Pesticides, aliphatic hydrocarbons and aromatic, PAH, heavy metals	Pesticides
Rubber production plants	Lead, tetrahydrofuran
Tanneries	Chrome
Polystyrene production plants	Styrene
Gas stations, transport bases, service stations vehicles, parking lots	Aliphatic hydrocarbons, mineral oils, gasoline, PAHs
Means of production and repair of means of transport, engine production	Aliphatic, polycyclic ring hydrocarbons aromatic hydrocarbons, heavy metals

15.3. BIOLOGICAL METHODS OF REMEDIATION OF DEGRADED AREAS

Interest in bioremediation in the scientific community is constantly increasing. Unlike physicochemical methods, technology based on the use of living organisms and biological processes is environmentally friendly, relatively cheap, and effective (Gałązka, 2015; Wołejko et al., 2016; Xia et al., 2019). Numerous microorganisms and plants show the ability to remove or transform arduous soil contaminants in processes such as bioaccumulation, biosorption, biotransformation or biodegradation (Awasthi et al., 2019; Gałązka, 2015). The mechanism of binding toxic substances by microorganisms is primarily associated with the presence of functional groups on the cell wall, such as carboxyl, phosphate, hydroxyl or sulfhydryl (Awasthi et al., 2019). It is worth mentioning that soil microorganisms, unlike other organisms, are characterized by exceptional adaptability to adverse environmental conditions, which translates into the efficiency of metabolism of many substances found in soils (Gałązka, 2019). In soil reclamation, phytoremediation is also a promising strategy, i.e. the use of plants, specifically their ability to accumulate pollutants in aboveground and underground organs and the distribution of toxic substances in the rhizosphere zone (Kowalska and Łukaszyk, 2020). These organisms can also be used to stabilize the substrate and delay the erosion process. Pollutant neutralization by appropriate plant species can occur through: phytodegradation, phytovariation (phytoparation), phytoextraction (phytoaccumulation), phytostabilization, phytostimulation (Gałązka, 2015). It is worth emphasizing that the main goal of bioremediation is not to neutralize pollution at all costs, despite the benefits of economy and simplicity of the process, but to reduce the risk of negative effects of toxic substances on the environment and thus protecting human health (Wołejko et al., 2016).

Biological soil remediation can be carried out in the place of contamination (*in situ*) by bioextraction, bioventilation, or cultivation of the soil outside the area (*ex situ*), in heaps, bioreactors or also with the use of various agrotechnical treatments, such as landfarming (Gałązka, 2015; Wydro et al., 2015). The selection of an appropriate reclamation method is undoubtedly related to both the activity of the organisms involved in the process and the characteristics of the cleaned soil (Nowak, 2008). The speed and effectiveness of the bioremediation process is determined by a number of factors (Fig. 15.2) (Wołejko et al., 2016), depending on the nature of the pollution and the structure of the soil environment and its physicochemical features. The ability to remove or immobilize xenobiotics by microorganisms depends primarily on the concentration, bioavailability, and mobility of toxic substances, as well as soil parameters such as pH, humidity, content of nutrients, redox potential and sorption capacity (Gałązka, 2015; Krosowiak et al., 2008; Rigoletto et al., 2020). For areas with a high degree of cohesion, e.g. clays, dusts or loamy sands, additional treatments are required to increase their permeability. In the case of *in situ* bioremediation, climatic conditions are also important from the point of view of the process, in particular temperature, which has a great impact on the metabolism of microorganisms (Gałązka, 2015).

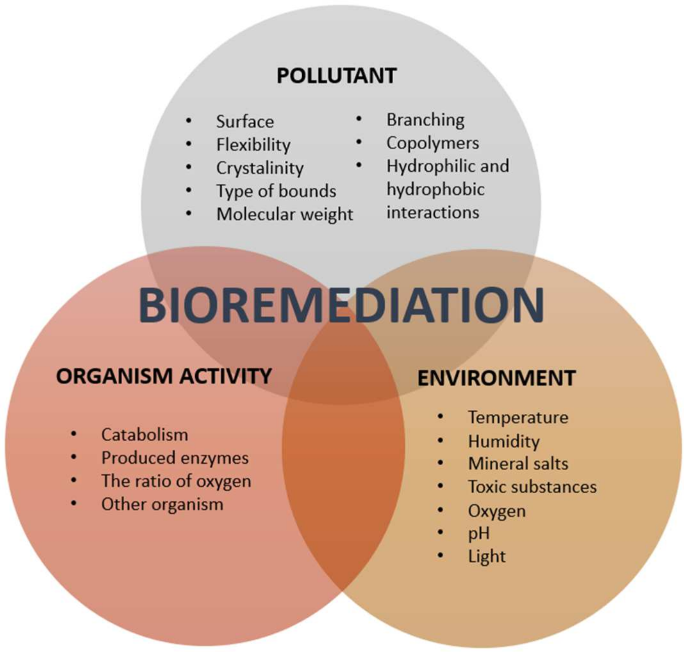

Fig. 15.2. The main factors determining the course of bioremediation (Wołejko et al., 2016)

An important problem that accompanies the proper course of the biological soil purification process is the presence of hardly degradable pollutants and their rapid accumulation in the environment (Wołejko et al., 2016). The rate of biological degradation may also be limited due to the low concentration of substances necessary for the development of soil microorganisms. Monitoring of natural processes of spontaneous soil cleaning, i.e. basic bioremediation, may not be sufficient to achieve a clear improvement in the quality of the area undergoing reclamation (Gałązka, 2015; Kołoczek and Kaszycki, 2005). Therefore, the need to modify bioremediation techniques by introducing materials and preparations (microbiological, enzymatic, organic) aimed at increasing the efficiency of degradation of compounds negatively affecting soil quality is justified (Vasilyeva et al., 2010; Wołejko et al., 2016). Nevertheless, this type of project must be preceded by an in-depth analysis of the substances applied and an assessment of the risk of far-reaching effects to avoid secondary pollution. The goal of targeted bioremediation support is primarily to increase the efficiency and effectiveness of soil degradation by using, inter alia, stimulation of the autochthonous population of microorganisms as a result of the supply of nutrients and oxygen, and the introduction of new active microorganisms capable of decomposing pollution in the degraded area (Janiszewska et al., 2017; Wydro et al., 2015). Modifications to basic bioremediation, including monitoring of the free biological distribution of soil contaminants, have led to the emergence of new types of biological remediation methods. These include: biostimulation, bioaugmentation, electrobioremediation, and rhizosphere bioremediation (Gałązka, 2015; Waraczewska et al., 2018).

Biostimulation is a technique of stimulating the native population of microorganisms in a given area, which aims to accelerate bioremediation. It includes all treatments aimed at eliminating factors limiting the natural course of biological neutralization of soil contaminants, such as low oxygen concentration, water deficit, unfavorable pH and temperature, or nutrient deficiency. An important element of planning the appropriate biostimulation strategy is conducting preliminary laboratory tests, the results of which are the foundation for making further decisions on specific solutions. The scope of research should take into account the type and structure of impurities and enable the definition of physicochemical properties of the contaminated area. However, it is found that all bioremediation processes occur faster under aerobic conditions. One of the strategies used in biostimulation is bioventilation, which allows for the removal of volatile forms of toxic compounds and accelerates the breakdown of heavier fractions. Other techniques to oxygenate the area include the use of a dilute hydrogen peroxide solution or mechanical soil loosening (Gałązka, 2015, Wołejko et al., 2016;).

Another type of engineering bioremediation is **bioaugmentation**, meaning an increase in the population of microorganisms found in a degraded area. The application of selected microorganisms showing the ability to biodegrade xenobiotics occurs when endogenous organisms do not show the desired catalytic activity. New microorganisms introduced into the soil must meet the following criteria:

- they cannot be pathogenic to plants and animals;
- they should not show antibiotic resistance;
- they should not produce toxins;
- should not participate in the exchange of genes encoding undesirable features;
- it is desirable that they be competitive with the indigenous population.

For some specific contaminants, it is not always possible to acquire microorganisms that will be able to break them down. A promising solution in this situation may be the use of genetic engineering techniques that allow for the construction of specialized strains synthesizing the appropriate enzymes. The limitation of such behavior is the risk of uncontrolled transfer of modified genes to other organisms, which is associated with the need for detailed research in this field (Gałązka, 2015).

Another option to improve the biological efficiency of land remediation is to use an electric field. **Electrobioremediation** involves a number of methods bringing together chemical and electrokinetic phenomena with simultaneous microbiological activity. The use of an electric field causes a faster, controlled flow of the electrolyte solution through the pores of the solid, which results in the creation of optimal conditions for the decomposition of substances susceptible to this process. Technology requires the selection of appropriate additional compounds that are media and electron acceptors (Gałązka, 2015).

In the case of phytoremediation, a significant effect of microbial activity in the root zone on the development and resistance of plants is observed. The rhizosphere, i.e. the part of the soil surrounding the root and bark cells is a habitat for many microorganisms, especially symbiotic papilloma bacteria of papilionaceous plants, mycorrhizal fungi and organisms that activate plant growth. This zone is characterized by a large number of metabolic changes, and thus an increased rate of biodegradation of pollutants. This phenomenon is used in **bioremediation methods in the rhizosphere**, due to the close interaction of plants, soil and microorganisms present in it (Gałązka, 2015).

Today's bioengineering is able to provide many solutions aimed at effectively healing degraded areas. It should be emphasized, however, that bioremediation is an interdisciplinary issue and includes disciplines such as biochemistry, microbiology, environmental engineering, process engineering, and ecology. The assessment of the effectiveness and possible effects of different remediation techniques should be preceded by meticulous research and projects on a larger scale, going beyond the laboratory environment (Wołejko et al., 2016).

15.4. NATURAL USE OF SEWAGE SLUDGE AND OTHER BIOWASTE

The proper waste management is the subject of interest for a number of both governmental and non-governmental organizations. This issue has been raised by many supranational institutions on an international scale for several decades. Appropriate waste management influences environmental protection and human health. In addition, it reduces the negative effects of misuse of garbage and ensures the efficiency of waste utilization by enterprises and municipalities (Żurawiecka and Kocia, 2019). In the case of biodegradable waste, which includes sewage sludge, high hopes are associated with its natural use. The ongoing modernization and expansion of infrastructure in the field of sewage disposal and treatment is a response to the need to adapt individual countries to the EU level. However, the amount of generated sewage sludge is steadily increasing, as the technology enabling complete removal of the sewage treatment product from the environment has not yet been developed (Bień et al., 2011; Miksch and Sikora, 2012; Singh and Agrawal, 2008;). The composition of sewage sludge is variable, it depends primarily on the properties of treated sewage and its treatment processes (Miksch and Sikora, 2012). The strategy for managing this waste is based on the properties of sludge, the scale of the threat to the environment and in legal terms, on applicable laws and regulations specific to this issue. Figure 15.3 shows the main directions of sewage sludge management in Poland (Bień et al., 2011).

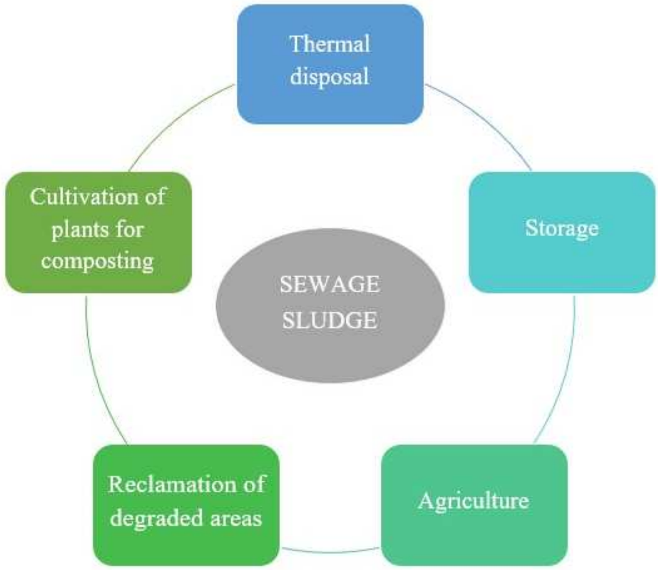

Fig. 15.3. Main directions of sewage sludge management in Poland (Bień et al., 2011)

In Poland, there is a gradual departure from the storage of sewage sludge, due to the problem of uncontrolled greenhouse gas emissions to the environment and the movement of pollutants to leachate. However, methods of thermal treatment of this waste have gained in importance (Kacprzak et al., 2017). Combustion processes, however, are expensive and their use is associated with the emission of harmful gases and dusts to the environment (Miksch and Sikora, 2012). Sewage sludge can be successfully used for soil fertilization and remediation if it meets the relevant hygiene and sanitary criteria (Hamdi et al., 2007). This solution is economically profitable, environmentally friendly, and fits in with the assumptions of the circular economy. This has been proven by some countries, such as Norway, which already in 2008 reported 80% of biosolids recycled for agricultural land or green areas. Dehydrated sludge, depending on the stabilization processes, contains about 50-70% organic matter and 30-50% mineral components, 3.4-4.0% N, 0.5-2.5% P and large amounts of other components nutrients. The organic matter of the sludge undergoes rapid mineralization due to the relatively low content of lignin-cellulosic compounds. Sewage sludge also contains elements such as nitrogen and phosphorus, which is a high fertilizer potential for plants. However, the wastewater treatment product may also include toxic substances (such as heavy metals), PAHs, PCBs, adsorbed halogenated organisms (AOX), pesticides, surfactants, hormones, pharmaceuticals, nanoparticles, and others. In addition, there are pathogenic organisms that pose a threat to human, animal and plant health (Kacprzak et al., 2017). In connection with the above, the introduction of sewage sludge to soils is associated with carrying out their thorough analysis, processing and stabilization (Miksch and Sikora, 2012).

Sewage sludge treatment is aimed at reducing their crease, limiting their mass and volume, and eliminating pathogenic organisms and parasites. The choice of the appropriate method of sludge transformation depends on the quality of the sewage, the size of the treatment plant and its location, the amount and physicochemical properties of the sludge, and the foundation conditions of the facilities and economic aspects (Miksch and Sikora, 2012). Table 15.2 presents methods of sewage sludge treatment together with the effect of their application (Miksch and Sikora, 2012).

Table 15.2

Methods of sewage sludge treatment and the effect of their application (Miksch and Sikora, 2012)

Treatment process	Results
Thickening	Sludge volume reduction
Stabilization (aerobic or anaerobic)	Reducing the amount of biodegradable organic matter; limiting the level of pathogens and the sludge's ability to rot; limiting the emission of odours
Lime stabilization	Increasing the pH of the sludge, reducing the number of pathogenic organisms and the ability of the sewage sludge to rot, increasing the dry matter content in the sludge
Conditioning	Increasing the dehydration capacity, the possibility of increasing the dry matter without increasing the content of organic substances; disinfection (in the case of thermal conditioning)
Drainage	Increasing the dry matter concentration by removing water; reduction of sediment volume
Composting	Decreasing biological activity; transformation of the sediment into a kind of humus
Drying	Sludge disinfection, significant reduction of odour production and biological activity

Sludge stabilization can be carried out in both aerobic and anaerobic conditions. The selection of the appropriate method is related to the infrastructure of the treatment plant and the economics of the technological solution. It is assumed that stabilization in aerobic conditions is profitable for small facilities, due to low input costs and high energy consumption of the process. However, in medium and large treatment plants, and especially in those where sedimentation is used in primary settling tanks, anaerobic digestion is commonly used (Miksch and Sikora, 2012).

Much like sewage sludge, other biodegradable waste can be treated. The use of biowaste in agriculture and land reclamation is an effective way to recycle organic matter and its valuable components. However, the use of such materials needs to be harmonized in legal terms, which is one of the main demands of the European Commission. The divergence of fertilizer rules and standards prevents the efficient production of organic fertilizers and their use in bioremediation (Mosqura-Losada et al., 2019). An attractive solution may also be the natural use of biochar, which is a solid produced by pyrolysis of biomass at high temperature in limited or conditions without oxygen. Its application is associated not only with the improvement of soil quality and structure, but also with the adsorption of pollutants, due to its porous structure (Chen et al., 2019).

As in the case of sewage sludge, prior to the natural use of biowaste, a thorough analysis and pre-treatment should be carried out, due to the risk of secondary soil pollution. For some substances, such as manure, composting and anaerobic digestion can be used as pre-treatment (Sosnowski et al., 2003). Recently, the method of anaerobic co-digestion of raw materials has been particularly popular. Anaerobic co-digestion is defined as a combination of decomposition of different types of substrates to achieve greater biogas yield. The combined processing of several wastes is also economically advantageous because it is possible to obtain materials from one source, e.g. a household (Cuetos et al., 2011; Wang et al., 2012).

A useful parameter to determine the possibility of using biowaste in agriculture and remediation is the ratio of carbon to nitrogen, which characterizes not only the soil subjected to reclamation, but also the raw materials used for this purpose. In the humus level of arable soils, the carbon content is usually 10 times higher than nitrogen, while in the case of very biologically active areas the C:N ratio is close to 8. Depending on the type of soil contamination, a different C/N value is observed. For example, in the case of organic pollutants such as PAH, a high carbon content is observed, and a C/N value greater than 45 means heavily degraded soil (Bielińska et al., 2012). Plant cover or its total disappearance reflects the state of soil degradation, however there is no clear relationship between the content of hydrocarbons in soil and plant vegetation. A constantly polluted area can be completely devoid of vegetation, with a relatively low content of hydrocarbons. The carbon to nitrogen ratio is therefore a better indicator of soil degradation than the hydrocarbon content (Gałązka, 2015). Knowledge about the carbon and nitrogen content of the contaminated area and the waste that is intended to be used therefore allows an assessment of the possibility of introducing the substances concerned into the soil. Table 15.3 shows the C/N quotient of selected organic materials (Sadecka and Suchowska-Kisielewicz, 2016).

Table 15.3

C/N value or exemplary organic substrates
(Sadecka and Suchowska-Kisielewicz, 2016)

Substrate	C/N	Substrate	C/N
Paper	170-800	Kitchen waste	12-20
Scobs	200-500	Green waste	10-25
Wood	700	Fresh grass	12-20
Bark	100-130	Legumes	18-20
Straw	80-100	Non-legume plants	11-12
Leaves and weeds	90	Manure	18
Maize cobs	40-80	Poultry manure	15
Hay	40	Food waste	15

15.5. POSSIBILITIES OF SOIL BIOSTIMULATION USING ORGANIC WASTE

The addition of biowaste in remediation of contaminated areas can bring significant benefits, such as soil enrichment with carbon and nutrients, reduction of acidity, promotion of immobilization of heavy metals, as well as the growth and development of organisms inhabiting this environment (Trebugova and Koptsik, 2019). In addition, compared to the use of inorganic fertilizers, the application of substances such as manure, compost or sewage sludge can maintain organic matter in the soil and improve the water retention capacity, which translates into increased microflora activity. The right level of organic compounds improves soil properties such as bulk density, permeability and porosity. The use of biowaste is also supported by the fact, that they are easily available and often are by-products of industrial activities (Hamid et al., 2020). However, in most developing countries, degradable organic matter from waste stored in the open is subject to uncontrolled, aerobic, or anaerobic degradation. These non-designed landfills cause the fine organic matter to mix with the percolation water to form a leachate that can endanger the environment. Therefore, in the face of intensive protection of natural resources and energy, recycling of organic waste is an extremely important issue (Padmavathiamma et al., 2008).

The use of biowaste for soils contaminated with petroleum compounds has turned out to be a success. Agamuthu et al. (2013) showed that the addition of sewage sludge and cow manure increases the biodegradation of grease by 82% for sludge and 94% for manure, respectively, compared to the control sample. This was probably due to the presence of an additional organic substance, which increased the ability of local organisms to degrade toxic compounds. In addition, it was found that the addition of liquid manure improved the physicochemical properties of the soil, thus enabling rapid adaptation of microorganisms to the impurities contained in it. In turn, Joo et al. (2007) showed a positive effect of composting diesel degraded soil together with food waste. In this case, petroleum hydrocarbons were degraded by 80% in just 15 days of the process. Joo et al. (2008) in later studies also presented the possibility of an improved composting process, namely in the presence of the *Candida catenulata* strain, which is characterized by a high ability to emulsify and degrade harmful hydrocarbons. The conducted experiment also brought beneficial effects, and the authors point to the need for further exploration of the subject in future experiments. Van Gestel et al. (2013) also composted soil contaminated with diesel oil. Researchers used plant, fruit and garden waste for this purpose. Their reduction in diesel concentration could have been the result of not only degradation, but also the volatilization of pollutants and / or adsorption on organic substances. The issue of adsorptive bioremediation of oil-contaminated soil has been further explored in the work of Vasilyeva et al. (2010). The authors suggest using a mixed adsorbent (ACD) composed of activated carbon and diatomite in this area. The introduction of this substance in combination with a biopreparation consisting of *Pseudomonas putida* B-2187 and *Rhodococcus* *erytropolis* bacterial strains resulted in a significant (2-7-fold) increase in the degree of degradation of petroleum compounds compared to controls. The use of adsorbent resulted in a significant reduction of soil phytotoxicity.

In the case of heavy metals, which belong to inorganic pollution of the environment, the addition of organic waste primarily allows for the immobilization of toxic elements. Biomaterials exhibit adsorptive, electrostatic, complexing and also allow metal precipitation. Their application to the soil has a positive effect on the pH, redox potential and soil sorption capacity, preventing hydrolysis and oxidation processes that determine the solubility and bioavailability of micro- and macroelements. Nowak et al. (2010), based on several column, vase and field tests conducted and confirmed the beneficial effect of sludge application for sewage on parameters of degraded soils, contaminated with heavy metals. Regardless of the species of the plant (grass, sunflower, willow, pine, birch, beech, alder), an increase in pH, sorption capacity, the number of microorganisms and elements: carbon, nitrogen and phosphorus has always been observed. Fijałkowski and Kacprzak (2009) had similar observations, showing a positive effect of the addition of sewage sludge on the physicochemical parameters of the soil from the area of the steel industry. However, it should be mentioned that the addition of a wastewater treatment product and other organic waste can lead to the release of organic acids, which lead to acidification of the environment, thereby increasing the mobility and biological availability of metals. Therefore, the effect of introducing biosolids on the mobility of elements depends on the production of dissolved organic carbon and organic acids (Hamid et al., 2020). In addition, there is a risk of introducing antibiotic resistance genes into the soil along with sediments. Xie et al. (2016), in their work, indicate that with increasing abuse of antibiotics by humans, the amount of these substances in wastewater will increase. Therefore, the addition of sediments to soils should be preceded by their detailed analysis in this respect.

Biochar is a particularly documented organic additive used in the bioremediation of areas contaminated with heavy metals. This compound contains oxygen functional groups, that enable metal complexation (Hamid et al., 2020). Figure 15.4 shows the reduction and immobilizing properties of biochar on the example of chromium (Xia et al., 2019).

The mechanism of binding impurities is similar as in the case of activated carbons, however, greater biochar affinity for soil organic matter causes carbonizates to have better complexing properties, as well as exhibit the ability to exchange sorbents interchangeably (Medyńska-Juraszek, 2016). Due to its porous structure, biochar can also be a carrier of microorganisms introduced into the soil. Wu et al. (2019) in their study on bioremediation of soil contaminated with cadmium, developed a preparation consisting of biochar and plant growth promoting bacteria, SNB6 strain. The experiment also used vetiver grass (*Chrysopogon zizanioides L.*), which has the ability to accumulate metals. The authors showed that the isolated SNB6 bacterial strain was successfully fixed on the biocarbon and successfully colonized at the rhizosphere interface. The use of a novel biochemical material, consisting of biochar and bacterial strain SNB6 enabled significant accumulation of cadmium by *C. zizanioides* and improved soil biochemical properties. In addition, an increase in the number of microorganisms and an increase in enzyme activity has also been observed. It should be remembered, however, that the addition of biochar in acidic soils may be helpful in limiting the bioavailability of metals, but its effect on neutral or alkaline soils may be slow (Hamid et al., 2020).

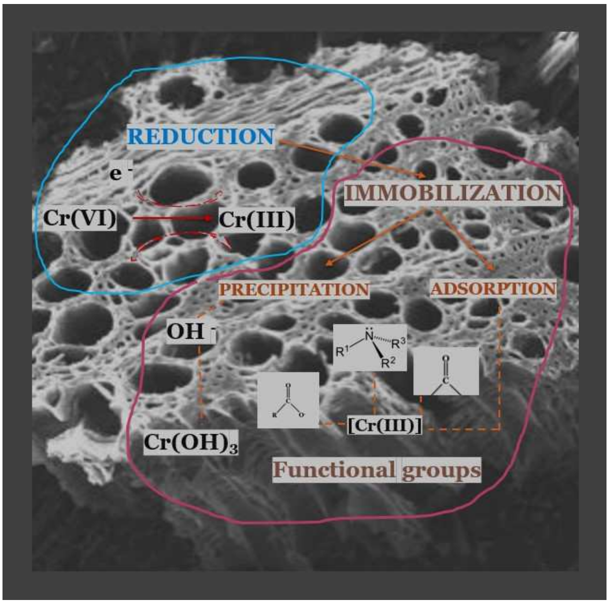

Fig. 15.4. Reduction and immobilization of chromium by biochar (Xia et al., 2019)

The combination of several organic wastes and specialized organisms can also be used to degrade pesticides, e.g. atrazine. This compound belongs to persistent organic pollutants, is characterized by extremely low susceptibility to natural decomposition and has a long half-life. Chen et al. (2019) developed an innovative fertilizer consisting of cow manure, biochar, poly (γ-glutamic acid) and *Arthrobacter sp.* DNS10 atrazine degrading strain. The analysis of the data obtained by the authors shows the high potential of the proposed method in remediation of contaminated soil, where the percentage of atrazine removal can reach up to 95%.

The application of organic additives to the soil in the form of biowaste also improves the quality of soils exposed to adverse abiotic factors. Mosquera-Losada et al. (2019), to remediate acidic Galician soils in northwester Spain, used fertilizers based on previously stabilized and sanitized waste, such as sewage sludge, animal by-products, quicklime, kindling, chicken manure and wood ash. The results of this experiment showed that high doses of biowaste fertilizers improved soil fertility, increased pasture production, and optimized botanical composition. The beneficial effect of fertilizer in the form on the cultivation of plants and physicochemical conditions of soils was also noticed by Liu et al. (2010). The authors examined the effect of chemical fertilizers, straw and manure on soil properties in northwestern China. The areas there are characterized by low hydration and low yields. The use of manure, straw and inorganic fertilizer resulted in an increase in soil fertility and microbial activity. The obtained results suggest that proper management of available biowaste and increasing the number of research in this area can significantly improve the quality of many degraded areas.

15.6. CONCLUSIONS

In the face of unavoidable changes in the natural environment as a result of anthropogenic activities, the protection of Earth's resources and proper waste management is extremely important. A radical shift towards ecological solutions in the field of land treatment is a rational approach not only in environmental but also in economic terms. Land bioremediation is a topic that is still explored due to the unlimited possibilities offered by the use of living organisms for purification processes, also obtained through genetic engineering. Stimulation of soil microbiological activity by the addition of organic matter in the form of biowaste is considered not only because of the reclamation of the degraded area, but also because of the possibility of reuse of waste, in accordance with the principles of circular economy. Stabilized sewage sludge present in the soil and compounds such as manure, crop residues or discarded food can have a positive effect on the physicochemical properties of the environment and mobilize microorganisms to faster degradation. What's more, materials with a properly porous structure, e.g. biochar, can be used as adsorbents for toxic compounds, but also as carriers for strains participating in remediation. The positive effect of bioremediation methods with the use of biowaste is observed in the case of extremely onerous pollutants, such as petroleum compounds, including PAHs, heavy metals, or herbicides. In addition, the application of organic substances improves soil fertility and yields. Nevertheless, the limitation to the use of such biowaste use is the risk of secondary pollution of the environment when these compounds are not safe in terms of hygiene and sanitation or contain toxic substances and antibiotics. Therefore, each decision to introduce sewage sludge and other biodegradable waste should be preceded by a thorough analysis, both physicochemical and microbiological, even at the genetic level.

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Part IV

Circular economy concept

Chapter 16

Circular economy guide

Magdalena MADEŁA

16.1. INTRODUCTION

In recent years, there has been a strong need to apply sustainable economic development good for the environment. Non-sustainable use of natural resources, loss of biodiversity, poor management of freshwater resources, overuse of land, increasing urban air pollution, ocean pollution and climate change are serious environmental problems. Therefore, as consumption, a better standard of living and a rapidly human population growth, the concept of a circular economy is becoming increasingly important. This model represents an environmentally sustainable direction of change, with the economy moving away from consumption, while extending the life and use of products and materials. Theoretically, a circular economy introduces significant economic and environmental benefits and should therefore replace a linear economy rather quickly, but in practice, the linear model still dominates the economy.

The aim of this chapter is to present the most important conditions and assumptions of the circular economy in order to improve the understanding of its concept. In the beginning the reviews were made the different available definitions the circular economy and the two material cycles are described: technical and biological, taking into account the economic and environmental aspects. This new economic model presents a transition from a consumption and disposal-based linear model to extending the life and use of products and materials. The circular economy model moves towards reuse, recycling of products and materials and renewable resources.

16.2. THE CIRCULAR ECONOMY

In recent years, there has been a worldwide need for a transition to an environmentally friendly industrial and economic development. This is why the concept of the Circular Economy has emerged. A number of authors have provided different concepts for defining the circular economy. Here are some of these definitions (Fig. 16.1):

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"Circular economy is an approach that would transform the function of resources in the economy. Waste from factories would become a valuable input to another process – and products could be repaired, reused or upgraded instead of thrown away" (Preston, 2012).

"A circular economy is an alternative to a traditional linear economy (make, use, dispose) in which we keep resources in use for as long as possible, extracting the maximum value from them whilst in use, then recovering and reusing products and materials" (Mitchell, 2015).

A circural economy is the "production and consumption of goods through closed loop material flows that internalize environmental externalities linked to virgin resource extraction and the generation of waste (including pollution)" (Sauvé et al., 2016).

A circural economy "refers mainly to physical and material resource aspects of the economy – it focuses on recycling, limiting and re-using the physical inputs to the economy, and using waste as a resource leading to reduced primary resource consumption" (Zaboli, 2014).

Fig. 16.1. Definitions of circulation economy

An essential element in the sustainable, low-carbon, resource-efficient and competitive development of the economy is the transition to a circular economy. Currently, most of the resources are extracted, processed, used, and finally disposed of as waste. However, waste is usually disposed of by incineration or landfill. The idea of a circular economy is to maintain the value of materials and products as long as possible. This minimises the need for new materials and energy (Lee et al., 2017).

The circular economy (CE) economy model, based on the circulation of matter in the earth's ecosystem, makes it possible to exploit the inherent potential in products that can be reused in the same or different form in a new cycle at the end of one life cycle as waste.

This reduces the negative impact on the environment, providing an alternative to the linear economic model, which is based on the "take, produce, use and throw away" principle (Ellen MacArthur Foundation, 2015b).

The circular economy model is characterised by a closed structure, which presents a new approach to resource flows, as opposed to the linear economy model. Therefore, the transition to a circular economy requires a comprehensive approach and systemic thinking when designing and inventing closed-circuit products and services (Paradowska, 2016). The concept of circular economy is based on three main principles, which are presented in Figure 16.2.

A modern economy requires applying these three principles in order to reintegrate the economy into the cohesive system of our planet. When extending the definition of rules, it is important to remember:

- Preserving and enhancing natural capital by controlling non-renewable stocks and balancing flows of renewable resources.
- Optimising resource efficiency through the circulation of products, components and materials used at the highest utility throughout in the area.
- Increasing the efficiency of the system by controlling negative externalities such as: water, air, soil, and noise pollution; climate change; and negative health effects related to resource use (Ellen MacArthur Foundation, 2015a).

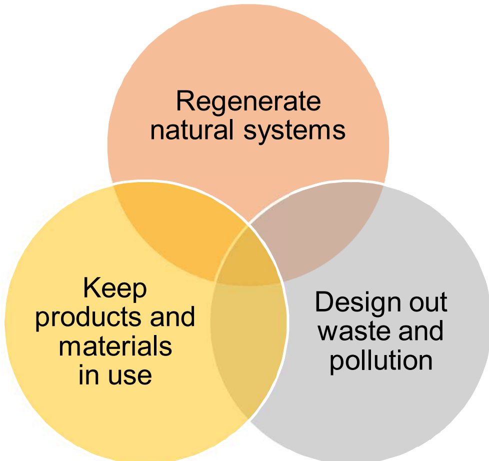

Fig. 16.2. The three main principles of a circular economy

In recent years, the concept of circular economy has been gaining more and more attention, as can be described as in the diagram (Fig. 16.3). It is divided into two material cycles: a biological cycle and a technical cycle. The aim is to maximise the use of clean, non-toxic materials and products designed to be easily maintained, reused, repaired, or renewed to extend their useful life, and then easily disassembled and processed into new products, while minimising waste at all stages of the extraction-production-consumption cycle.

The technical cycle includes the management of finite material stocks. These materials are recovered and mainly restored in the technical cycle. In the biological cycle there are the flows of renewable materials, whereas renewable (biological) substances are mostly regenerated in the biological cycle (Ellen MacArthur Foundation, 2015b).

The basic features describing the circular economy are shown in Figure 16.4. In a circular economy, it is necessary to proceed in such a way that waste does not exist. Biowaste is non-toxic, can be composted or digested and returned to the cycle. However, the man-made materials such as polymers, alloys and others can be recovered, refreshed, and improved with minimum energy input and maximum value retention. The key factor is diversity, which the economy needs. A balance is needed between the different scales of businesses to enable them to grow in the long term (Ellen MacArthur Foundation, 2015b).

It is assumed that the energy needed to drive the circular economy should be renewable in order to reduce resource dependency and increase the resilience of systems. In a circular economy, prices act as indicators that should reflect real costs.

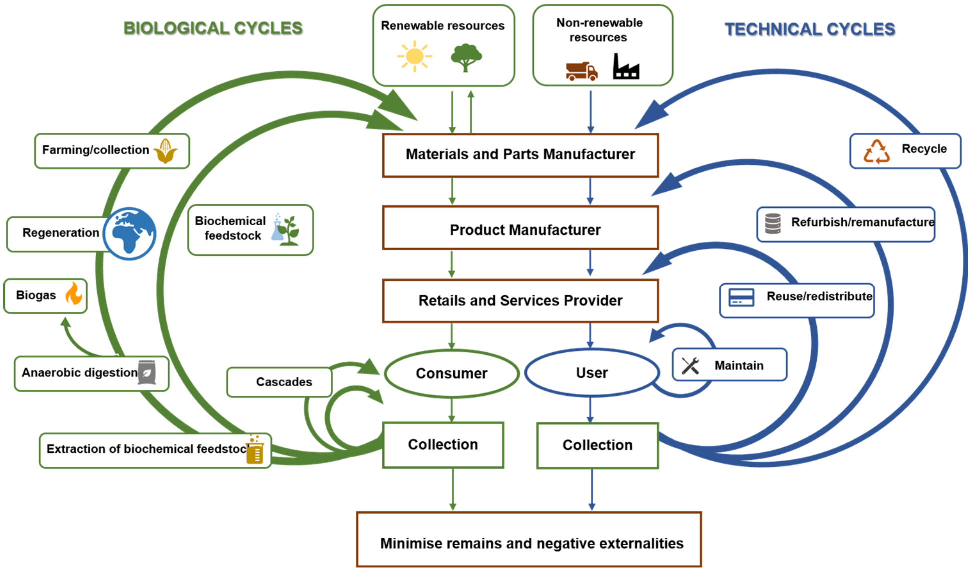

Fig. 16.3. Scheme of the circular economy (on the basis of Ellen MacArthur Foundation, 2015a)

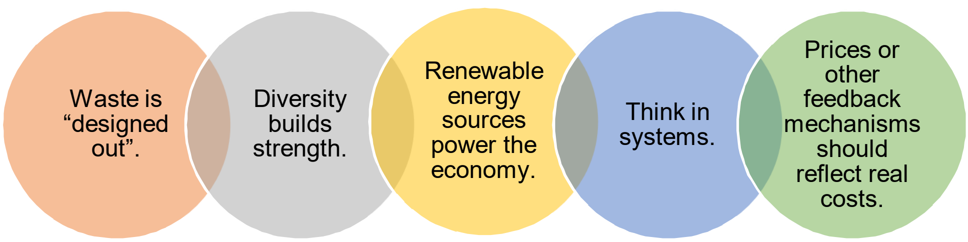

Fig. 16.4. The features describing a circular economy

An important aspect in the transition to a circular economy is that companies should begin changing their business model to a circular one. On the basis of the data analyses, a set of six actions has been developed to help entrepreneurs and governments to switch to a closed economy. These actions are called the ReSOLVE framework, which give entrepreneurs and governments the tools necessary to build strategies and initiatives. A diagram showing these activities is shown in Figure 16.5. These activities make it possible to increase the use of physical resources, extend their life and use renewable resources (Ellen MacArthur Foundation, 2015b).

REGENERATE

- Return recovered biological resources to the biospher
- Reclaim, retain, and restore health of ecosystems
- Shift to renewable energy and materials

SHARE

- Prolong life
- Reuse/secondhand
- Share assets

OPTIMISE

- Leverage big data, automation, remote, sensing and steering
- Remove waste in production and supply chain
- Increase performance/efficiency of product

LOOP

- Extract biochemicals from organic waste
- Digest anaerobically
- Recycle materials
- Remanufacture products or components

VIRTUALISE

- Dematerialise indirectly
- Dematerialise directly

EXCHANGE

- Choose new product/service
- Apply new technologies
- Replace old with advanced non-renewable materials

Fig. 16.5. The scheme ReSOLVE by Ellen MacArthur Foundation (Ellen MacArthur Foundation, 2015b)

Biodegradable waste is a broader concept than biowaste, as it includes industrial streams in addition to household waste and streams that produce similar waste. The biowaste management options include: prevention at the source, collection and separation, composting, anaerobic digestion, incineration, and landfilling. These activities often generate recycling and energy products (Manfredi, 2011). It is also possible to reuse biowaste as sorption materials (Madeła, 2021). This generally has positive environmental effects, depending on the recovery processes (Manfredi, 2011).

Figure 16.6 shows treatment methods, recovered products, energy recovery and related avoided products from the management of biowaste depending on its collection.

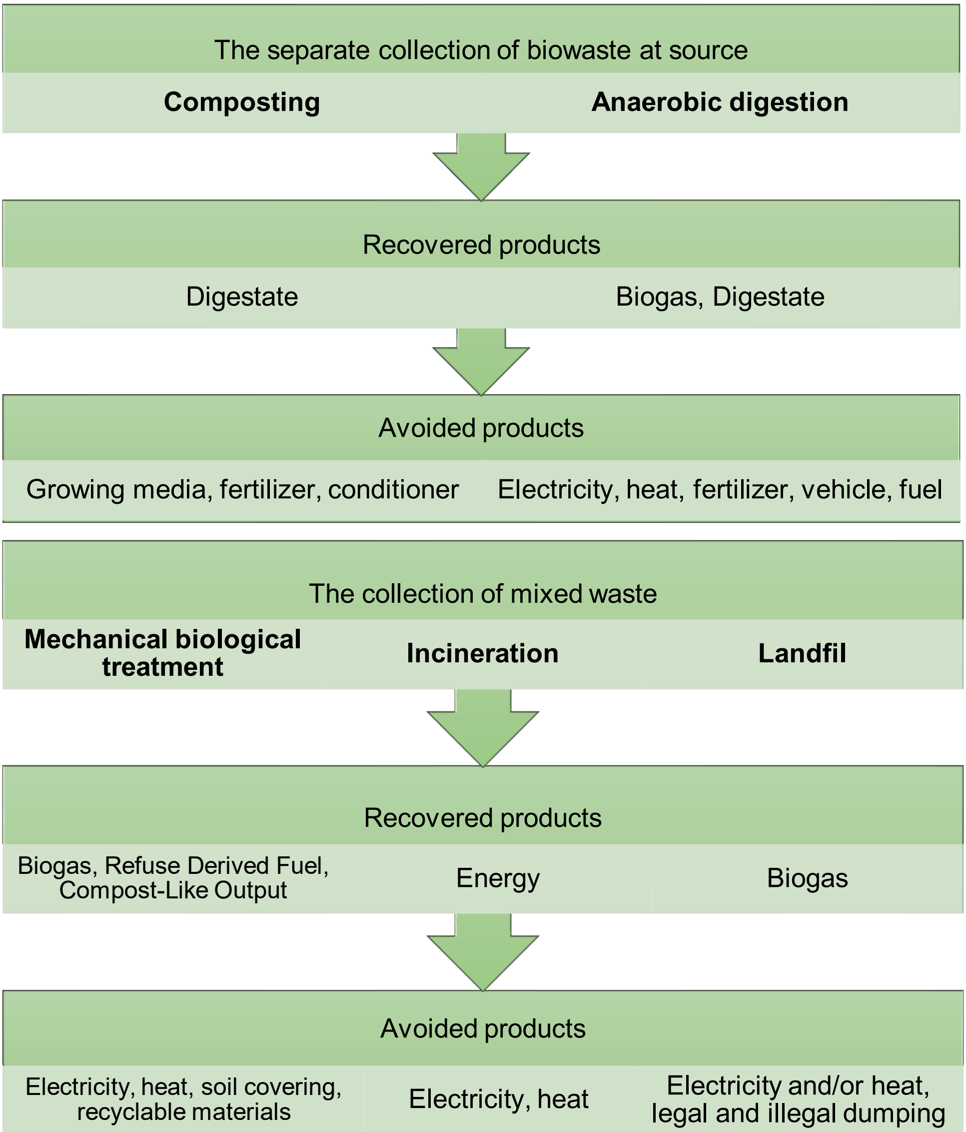

Fig. 16.6. Dependency scheme during the treatment of biowaste (on the basis of Manfredi, 2011)

The applying of waste should lead to the most resource-efficient method of dealing with it, and the decision-making process is easy, fast, and cost-effective. There may, however, be a need, in specific circumstances, to move away from the known practice for environmental reasons.

16.3. CONCLUSION

This chapter presents the principles of the circular economy and natural resource management. The circular economy is a complex concept taking into account a large number of components, characterised by a systemic approach that views the product as a reusable life cycle element. The increasing use of natural raw materials calls for attention to be paid to the potential for greater reuse of secondary raw materials. Paying particular attention to the increased use of renewable resources, it is therefore essential that resource management is already taken into account at the product design stage. In the framework of sustainable management, the values of products, materials and resources should be preserved in the economy for as long as possible in order to minimise the generation of waste. At the same time, a system for separate collection and treatment of waste should be implemented. An important element of the circular economy is the bioeconomy. The circular economy has economic, social, climatic, and environmental benefits.

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Chapter 17

Life cycle assessment (LCA) – application of the process

Rafał NOWAK
Elżbieta SPERCZYŃSKA

17.1. INTRODUCTION

LCA (life cycle assessment) is defined as an assessment of the life cycle from "cradle-to-grave". It assesses the effect of product or technology on the environment at all stages of their lifes. LCA is a methodology that supports planning, organization, and management, whose main purpose is the assessment of potential threats concerning environmental aspects. In other words, LCA is aimed at estimating the environmental risk of the technological process for the final effectiveness of the of the applied solution (Güereca et al., 2019). This methodology, regarding its nature, analysis quality, and data completeness, can be used as a tool supporting the decision-making process in the whole life cycle of the product or technology.

The aim of the study is to present general information on the life cycle assessment technique. Historically, the first mention of LCA methodology could be found in the study of Harold Smith, presented at the World Conference on Energy in 1963 (Klöpffer and Grahl, 2014). Research conducted by Smith focused on various chemical methods of energy production. One of the first companies interested in using these analyses in practice was Coca-Cola Company. In the late 1960s, the concern commissioned the study on each kind of liquid packaging for raw materials, expenditure, and energy used in the production process with environmental impact.

The development of LCA was also influenced by issues related to the formation of acid rains and the greenhouse effect. In 1993 Society of Environmental Toxicology and Chemistry (SETAC) has published the first LCA procedure "A code of practice", which gained widespread acceptance (Hauschild and Huijbregts, 2015). Over the same period the International Organization for Standardization (ISO) also started working for a standardization of the LCA process. As a result, ISO 1404x standards were developed. International Standard ISO 14040 defined LCA as, (ISO, 2006): "A life cycle assessment (LCA, also known as life cycle analysis, eco-balance) is a technique for a product-related

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estimation of environmental aspects and impact. LCA assesses every impact associated with all stages of a process from cradle-to-grave (i.e., from raw materials through materials processing, manufacture, distribution, use, repair, maintenance, and disposal or recycling).”

The concept of the LCA process is presented in Figure 17.1.

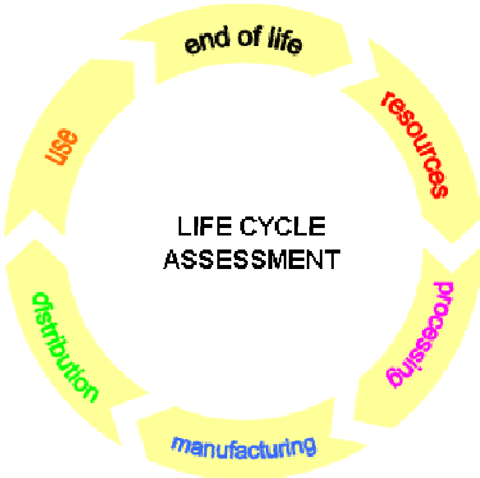

Fig. 17.1. Life cycle assessment (Brusseau, 2019)

17.2. STAGES OF THE LCA PROCESS

According to ISO 14040 LCA process consists of four stages (ISO, 2006):

17.2.1. PURPOSE AND SCOPE OF LCA ANALYSIS

This stage defines the context of the study, technical details, way to communicate the results, and the final addressee. It aims to demonstrate how great a part of the cycle of product/ technology will be estimated. Both the aim and scope should be clearly defined and coherent with the destiny as well as with end-use. The nature of the LCA process may cause that during the study redefinition of the aim and scope will be necessary.

17.2.2. LIFE CYCLE INVENTORY (LCI)

The second step is analysis of the data set of inputs and outputs. Collect and analyze data relating to inputs and outputs to/from the environment. These data are collected for each unit process specified in the product system. A statement is made of the number of materials and energy entering and leaving (by-products, emissions, waste) to/from a given process. Input data and output data are compiled for unit processes that are inside the product system boundary and contribute most to mass and energy flows and cause significant releases to the environment. This takes account of the flow balance of materials and energy in the product system in the context of interactions with the natural environment, used raw materials, and emission of harmful substances (including noise and odors). In this step, the data is assigned to individual unit processes (data allocation) and converted into a functional unit (e.g. per ton of product or product unit).

17.2.3. LIFE CYCLE IMPACT ASSESSMENT (LCIA)

The third stage is the most important part of the analysis. It allows for a detailed description of the impact of product/technology on the selected elements of the natural environment. It also allows for the numerical interpretation of this impact, including calculations of category indicator. In this way it is possible to estimate the volume of natural gas used for the production of the product or application of the unit process, and also estimate its effect on global warming as the result of this gas burning.

17.2.4. INTERPRETATION OF THE LIFE CYCLE

The last stage includes critical analysis, identifying the boundaries of the technology and its working conditions, decision criteria taking into account the costs and yield, presentation of optional solutions, as well as a presentation of the results and conclusions including graphical interpretation of the data. This stage fits completely with the aim and scope of the analysis. The relationship between various LCA stages is presented in Figure 17.2.

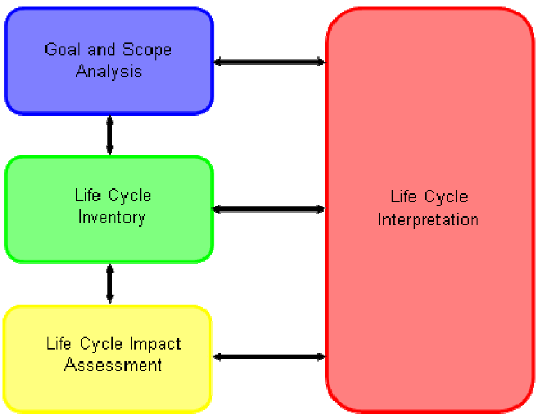

Fig. 17.2. Phases of LCA (ISO, 2006)

17.3. WHAT IS THE PURPOSE OF THE APPLICATION OF LCA?

The range of LCA is wide, and this method is used mainly for the determination of the real impact of various solutions on the natural environment, and consequently choosing the least harmful technology or product. And that applies to (Goermer et al., 2020):

- designing of products/ processes and/ or technologies;
- improvement of products and/ or technologies;
- comparative analyses of applied solutions;
- system of eco-labeling (process, product environmentally friendly);
- establishing rigorous pro-ecological standards;
- eco-product policy development;
- designing sustainable development strategies;
- waste management;
- marketing activities.

The use of LCA makes the decision-making process easier for companies and also leads to improving designed product/ technology concerning environmental protection. More and more economic players implement the technologies based on LCA within sustainable development, because this is a precondition to obtaining the ISO 14001 certificate (ISO, 2015). This leads to a revised life cycle of product/technology. The production process involving concept, production, packaging, and distribution finally includes application of the product concerning the effect on human health, use of natural resources as well as potential ecological threats, taking into consideration the quantity of water and energy used, generation of waste, and the CO_2 footprint.

The LCA method, especially extended with the functions of assessing individual scenarios and their optimization, can be used in the implementation of the idea of the circular economy (Lausselet et al., 2017). The concept of a circular economy assumes waste minimization at the product design level, and then covers the successive phases of the life cycle, up to the reuse or recycling phase. Such possibilities are offered by the LCA technique because it enables (Zarębska and Joachimiak-Lechman, 2016):

- determination of environmental loads on products and selection of the least resource-consuming and energy-consuming products from among them,
- assessment of the environmental impact of alternative ways of performing the same function by different product systems,
- comparison of production processes in terms of the production factors used,
- identification of potential environmental impacts of selected streams or unit processes and their comparison, presentation of the ratio of burdens to environmental benefits,
- reducing the amount of waste generated.

17.4. LCA FROM A STANDARISTION PERSPECTIVE

ISO 14040:2006 (ISO, 2006), ISO 14044:2006 (ISO, 2006), ISO 14045:2012 (ISO, 2012), ISO 14046:2014 (ISO, 2014), ISO/TR 14047:2012 (ISO, 2012), ISO/TS 14048:2002 (ISO, 2002), ISO/TR 14049:2012 (ISO, 2012), ISO/TS 14071:2014 (ISO, 2014), ISO/TS 14072:2014 (ISO, 2014), ISO/TR 14073:2017

(ISO, 2017), ISO/AWI TS 14074 standards have been assessed as guidelines in the European Union by the European Committee for Standardization (CEN). Additionally, Polish standards are based on these standards. They are:

- PN-EN ISO 14040:2009 – basic rules and instructions concerning LCA without detailed description of the LCA methodology (PKN, 2009);
- PN-EN ISO 14044:2009 – presents requirements and guidelines (old standards ISO 14041, ISO 14042, ISO 14043) (PKN, 2009);
- PN-EN ISO 14045:2012 – presents principles, requirements, and guidelines (PKN, 2012);
- PN-EN ISO 14046:2016-04 – water footprint – principles, requirements, and guidelines (PKN, 2016);
- PKN-ISO/TR 14047:2006 – presents examples of application of ISO 14042 (PKN, 2006).

17.5. ECO-LABELING

LCA as a tool of sustainable development is the basis for obtaining eco-labeling and environmental certificates. Eco-labeling impacts the perception of the trademark, value, and image of the company and also plays a role in increasing the competitiveness of the products compared to other companies' products. Eco-labeling is a recognizable logo that help facilities make more sustainable shopping decisions. It directly contributes to an increase in turnover and as a result increases the portability of the company. Labeling of ecological products and processes carries out the following functions: offers protection of the natural environment, provides information about ecological characteristics, encourage the companies to change the technologies they use to more environmentally friendly ones. It also carries out an educational function by expanding the knowledge on ecologically focused characteristics of the products (Thøgersen, 2010; Laso, 2017).

17.6. SOFTWARE FOR LCA ANALYSIS

For LCA analysis, more and more frequently specialized software is used. Often the software is developed by university specialists, societies, and organizations active in the field of environmental protection and provides support for acts in the area of environmental engineering and protection. The software is used for modeling and reporting life cycle (LCA) concerning environmental footprint. It includes databases that allow for the assessment of each of raw materials used and each process applied in production, at every stage, from the extraction of the resource to the end of the life cycle, taking into consideration the entire supply chain. The most popular tools (software) are: SimaPro, LCA Manager, OpenLCA, Umberto or GaBi Software (Iwaniuk, 2013; Lesiuk, 2012; Silva, 2019; Szamosi et al., 2020; Vervaeke, 2012).

Use of specialized software by companies contributes noticeably to not only protection of the environment and reducing the risk of its pollution, but also to (Iwaniuk, 2013; Lesiuk, 2012; Silva, 2019; Szamosi et al., 2020; Vervaeke, 2012):

- replacement of harmful components into the harmless ones,
- design changes of the products,
- develop environmentally friendly technologies,
- reorganization of production and distribution processes,
- improvement of business profits while maintaining the advantages of the products or technologies,
- establishing of new production standards,
- environmentally oriented management,
- change in branch perception (eco-labeling).

17.7. CONCLUSION

LCA is a life cycle assessment methodology, which allows for evaluation of the impact of the product or technology to the environment. It is an integrated approach to the environmental interactions and it includes a range of activities, from raw material extraction, through production, energy distribution, up to final utilization of the product.

In the company, LCA is simultaneously used both in accounting and environmental management. By implementation of the 14040 ISO standard, the following advantages are possible: sustainable development of enterprises, certification enabling the use of eco-labeling of products, as well as critical analysis of the projects. Use of specialized software assists in modeling and interpretation of the LCA analysis as well as in taking up sustainable actions favoring environmental management. LCA unambiguously contributes to broadening the awareness about the effect of product/technology in the perspective of the whole life cycle of a product. LCA standards are valuable guidance and standards that can help reduce your Product Carbon and Environmental Footprint.

The most important advantages of research using the LCA technique include: flexibility, interdisciplinarity, comprehensiveness, compatibility, and the result as a number. The main disadvantages of the LCA technique include: subjectivity, the fact that it's time-consuming and expensive, no spatial and temporal differentiation, incomplete data, and complexity of analyzes.

Despite the disadvantages presented, this method is becoming more and more important as a tool supporting environmental management all over the world. It is being constantly developed and improved, mainly in the direction of using it as a tool in the aspect of circular economy by extending its possibilities (e.g. adaptation with risk assessment) or combining it with other methods of environmental assessment (e.g. LCC – Life Cycle Costing).

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Chapter 18

Determination the criteria for selecting the best solutions for safe biowaste management in accordance with the requirements of closed-circular management, local market demand and the paradigm 'waste or a resource'

Lidia WOLNY

18.1. INTRODUCTION

The application of systems thinking is extremely important considering the transition to a circular economy concept in general. Transformation of waste into secondary raw materials ensures reorganization of linear material flows of a conventional economy into circular flows, where waste generation is excluded.

Circular Economy (CE) is an alternative model of the economy, which is based on closing the product's life cycle. In practice, CE is about maximizing use of the products and re-using them for the same or another purpose after the end of the life cycle. This makes it possible to maintain materials and resources in the economy as long as possible and to reduce landfill waste. Circular Economy was adopted by the European Union as a strategy for achieving its environmental goals and increasing the competitiveness of the EU economy. Poland has committed to transforming its economy into the CE, but it is only at the beginning of the road and has to deal with social, technological, legislative and financial barriers. Another important feature of a circular economy arises from one of the main subjects it addresses, namely waste, which can contain harmful or hazardous pollutants. In the case of sewage sludge, there is a risk associated with possible negative impacts to the environment by pathogenic substances, endocrine disrupters, heavy metals, and the accumulation of heavy metals in living organisms (Hollins et al., 2017; Ordinance of the Minister of the Environment, 2015).

The use of multi-criteria analysis methods in waste management usually comes down to the choice of the optimal solution in the waste management system

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specifically designated region and comparisons between each other's waste management technologies.

Criteria for environmental assessment may include: visual impact, air emissions, water discharge, ash discharges, human health, fauna and flora, site operations traffic socio-economic impacts, land-use and cultural heritage (Williams, 2006).

Environmentally sound waste management has been a cornerstone of European policies from the beginning, and significant progress has been achieved regarding the reduction of impacts from waste generation to the environment and human health (European Environment Agency, EEA, 2015).

The current development of research issues related to waste management in the world can be systematized in accordance with the existing hierarchy of waste management: avoiding processing, minimalization, material recycling, recovery (including energy) and neutralization.

The overall aim of the paper is to determine the waste management solutions in the context of a circular economy transition.

18.2. SOURCES, COMPOSITION, AND CHARACTERISTIC OF THE BIOWASTE

The description of biowastes can be made according to their potential and main source of their origin. Biowaste and residues include not only food waste, but also, for example agricultural, forestry, marine and animal derived residues. While disposal of some of these waste streams has historically been considered a challenge, with the aid of new technologies and tackling market barriers in the take up of well-established technologies, these waste streams are being rethought within the context of a closed circular economy, with waste streams re-categorised as either feedstock, raw materials, or energy.

The wastewater sludge is a large – tonnage waste with some specific characteristic, which depends on a wide variety of factors such as the seasons, the technology applied in wastewater treatment plants (WWTPs), the specificity of the source area of the influent, etc. On average, dewatered sewage sludge contains 50-70% organic matter and 30-50% mineral components (including 1-4% of inorganic carbon), 3.4-4.0% nitrogen (N), 0.5-2.5% phosphorus (P), and significant amounts of other nutrients, including micronutrients that could be recovered (Grobelak et al., 2019). On the other hand, sewage sludge is this kind of waste, which contains a number of harmful contaminants, such as toxic heavy metals, as well as organic toxins (e.g. polychlorinated biphenyls), pharmaceutical residues, pathogens and others (Tsybina and Wuensch, 2018; Vea et al., 2018).

Biowaste in the circular economy shall not be landfilled. It forms a resource for organic soil improvers, fertilisers, a growing media component and bio-based products instead. The circular economy concept offers a number of solutions to increasing amounts of biowaste and lack of resources by valorising biowaste. However, it is necessary to consistently address the environmental benefits and impacts of circular biowaste management systems (CBWMS). Various decision support tools (DST) for environmental assessment of waste management systems (WMS) exist (Vea et al., 2018).

18.3. FROM LINEAR TO CIRCULAR ECONOMY

The linear economy may be summarized as follows:

- take (the resources you need);
- make (profit and goods);
- dispose (of everything not needed, also the product at the end of its lifecycle).

A circular economy (Fig. 18.1) is restorative and regenerative by design and aims to keep products, components, and materials at their highest utility and value, at all times. The concept distinguishes between technical and biological cycles (Ellen McArthur Foundation, 2015).

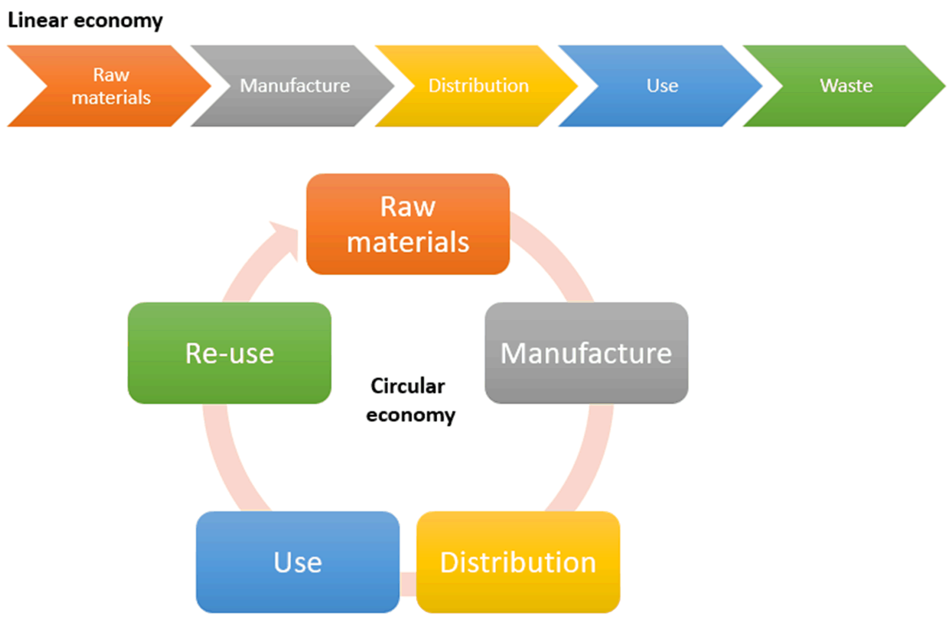


Fig. 18.1. Differences between the linear and the circular economy (according to Ellen McArthur Foundation, 2015)

Bio-based economies can be defined as: "technological developments that lead to a significant replacement of fossil fuels by biomass in the production of pharmaceuticals, chemicals, materials, transportation fuels, electricity and heat" (Sariatli, 2017). The very closely related concept of "bio-economy" usually focuses on the utilization of biomass in primary production processes in forestry, fisheries, and agriculture and increased valorization of raw materials used.

As opposed to the linear economy, the concept describes how to develop closed-loop technical and biological cycles by either recycling materials indefinitely with no degradation of their properties (the technical cycle) or returning materials to the natural ecosystem with no harm to the environment (the biological cycle). It tends to focus on an increased quantity of reused and recycled resources and overlook the quality of resource flows re-entering to the product cycle (Cullen et al., 2017; Thomsen et al., 2017).

The circular economy concept was chosen as a basis of analysis since it is one of the priority concepts of economic development underlying the current European policy in the field of environmental protection. To ensure the transition to a circular economy, the European Commission has developed an Action Plan for the Circular Economy, in which four key action areas have been defined (European Commission, 2015). In the case of sewage sludge, the study should be focused only on two key areas of this action plan: waste management and secondary raw materials.

However, Waste-to-Energy (WtE) can be attributed to different labels, such as ‘disposal’, ‘recovery’ and potentially ‘recycling’ for anaerobic digestion (as proposed by the WtE Communication, shown in Figure 18.2).

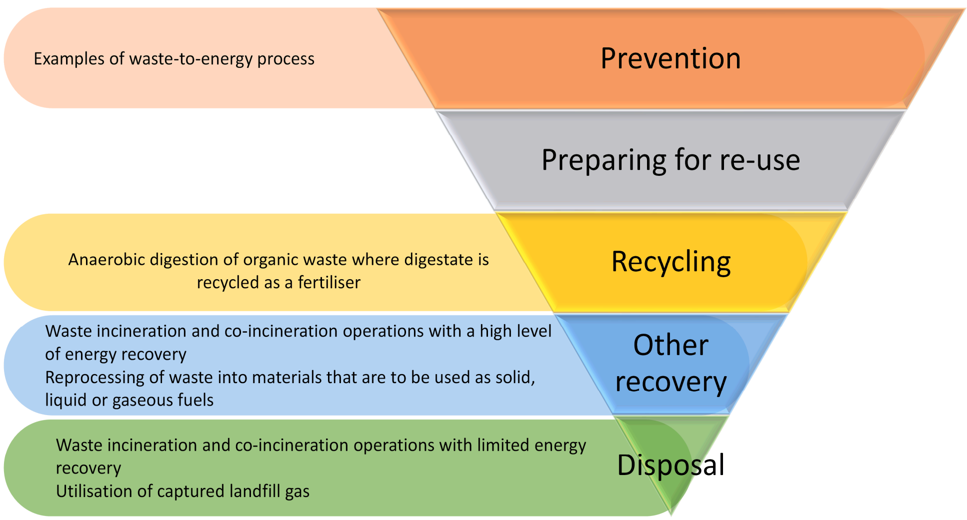
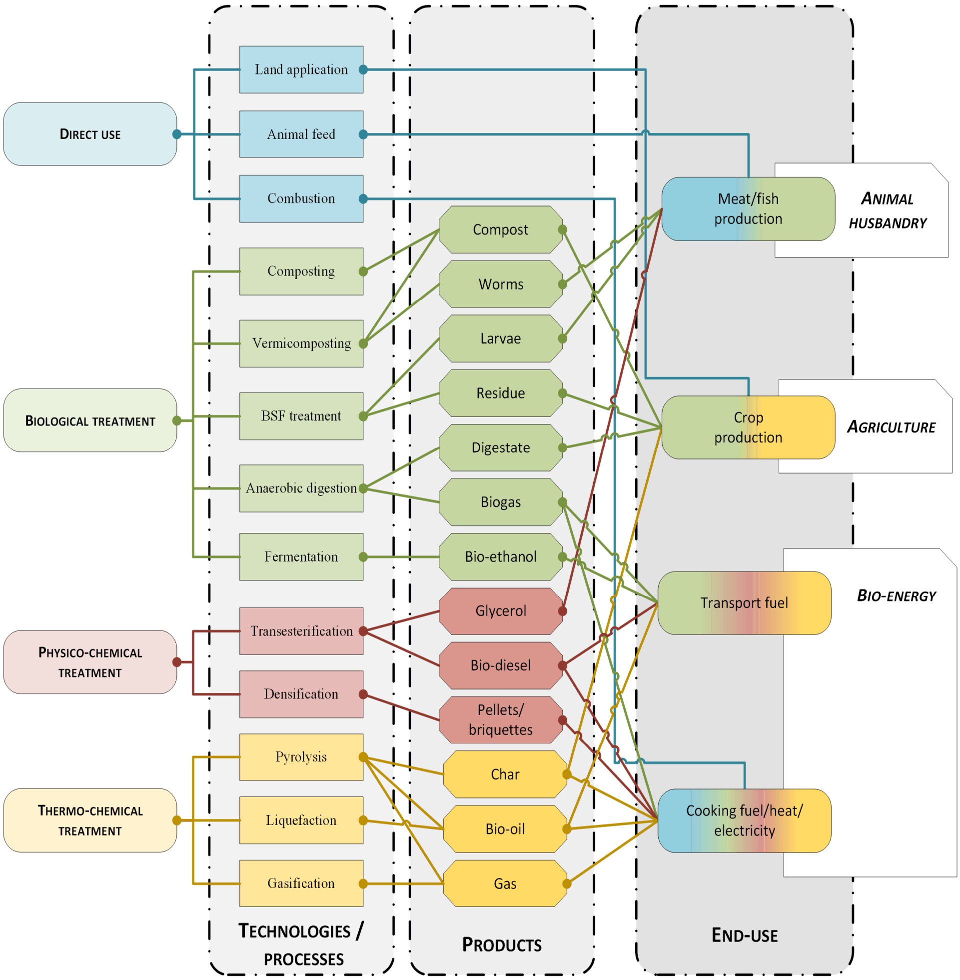


Fig. 18.2. WtE in the Waste Hierarchy (European Commission, 2017a)

Waste management and the recovery of secondary raw materials play a significant role in a circular economy. As it is specifically indicated in the action plan, the EU waste hierarchy should be applied so that the options that deliver the best environmental outcome are encouraged. Biological materials are to be returned to the natural metabolic cycles after the necessary pre-treatment, such as composting or digestion. Anaerobic digestion has been used to treat sewage sludge and agricultural wastes for many years and has also been developed for municipal solid wastes and industrial wastes. The process of anaerobic digestion of organic waste takes place in an enclosed, controlled reactor. The main aim of the process is to produce a product gas, rich in methane, that can be used to provide a fuel or act as a chemical feedstock.

An overview of biowaste treatment technologies/processes was presented in Figure 18.3.



BSF - Black soldier fly

Fig. 18.3. An overview of biowaste treatment technologies with the respective products generated from waste and their end-use (according to Lohri et al., 2017)

In this context, biowaste treatment technologies are defined as processes that convert discarded biowaste into new products with potentially some value. Treatment technologies for urban solid biowaste are grouped into four main categories:

1 – direct use,
2 – biological treatment,
3 – physicochemical treatment,
4 – thermochemical treatment.

Sustainable waste recycling requires a supply of adequate waste materials as input, and the market demand for the output products (Vergara and Tchobanoglous, 2012).

For biowaste, such markets will depend on the intended end – use of the outputs, which can be clustered into three-end-use groups: animal husbandry, agriculture, and bioenergy. Waste that cannot be avoided or recycled should be used to recover its energy potential, which is considered preferable to landfilling. The introduction of secondary raw materials into the economy is considered a positive factor that extends the security of supply. This would mean fewer risks connected to exposure to volatile raw material prices, as well as fewer risks connected to an unstable supply because of sudden natural disasters or changes in geopolitical situations. Nutrients are an especially important category of the secondary raw materials produced out of waste (Ellen McArthur Foundation, 2015; Cieślik and Konieczka, 2016).

The interest of European Union of Circular Economy involved the actions in this subject also in Poland. In September of 2015, Poland was prepared some comments, which European Commission was taken into consideration involving the CE plan.

Poland's comments included four issues as below:

- Crucial role of innovation of the European economy and a stronger link between the research and application;
- Establishment of a market for good quality secondary raw materials;
- Good quality of resources as a result of sustainable production and consumption;

Considerable potential of the service sector.

18.4. BEST ENVIRONMENTALLY MANAGEMENT PRACTICE

The best practices for the waste management phases and activities with the greatest circular economy potential concerning:

- establishing a waste management strategy;
- supporting waste prevention;
- promoting the reuse of products and preparation of waste for reuse;
- waste treatment, limited to operations enabling material recycling.

Table 18.1 summarised the main environmental burdens associated with different aspects of organic waste recycling, principally anaerobic digestion (AD) and composting, but also energy recovery via combustion (green waste).

Table 18.1

Main environmental impacts arising from organic waste recycling (European Commission, 2016)

Environmental aspects	Main environmental impacts
Separated organic waste collection	– Fossil resource depletion – Traffic congestion and noise – Odour nuisance – Pest nuisance
Infrastructure construction and maintenance	– Abiotic resource depletion – Fossil resource depletion – Land occupation
Machinery operations	– Fossil resource depletion – Global warming – Acidification – Photochemical ozone formation
Biogas leakage (composting and anaerobic digestion)	– Global warming (CH_4) – Acidification and eutrophication (NH_4)
Digestate and compost storage and application	– Acidification and eutrophication (NH_4) – Fossil resource depletion – Global warming potential (diesel CO_2 plus soil N_2O) – Avoided fertiliser manufacture and application burdens – Avoided global warming potential (soil carbon sequestration)
Energy recovery (biogas or biomass combustion)	– Acidification (NO_x and SO_x) – Photochemical ozone formation (VOCs and NO_x) – Human toxicity (particulates and polycyclic aromatic hydrocarbons) – Avoided fossil fuel combustion burdens
Extracted inorganic materials and combustion ash	– Landfill burdens

18.5. CRITERIA OF ASSESSMENT SUSTAINABILITY OF WASTE MANAGEMENT SYSTEMS

In 2015, 241 million tonnes of municipal solid waste were generated in the EU (Eurostat 2015). Of this waste, 40-60% was organic waste (Fava et al., 2015), representing a great challenge in terms of its management. However, at the same time, organic waste also constitutes a valuable resource as a component in the circular bioeconomy. The study concerns circular economy systems related to management of municipal biowaste as circular biowaste management systems

(CBWMS). Several decision support tools (DSTs) based on life cycle assessment (LCA) are currently available to assess the sustainability of waste management systems (WMS). These WMS-DSTs are specifically developed to analyse the performance of integrated WMSs from collection, treatment, and final disposal (Lokesh, 2018).

Recently, many authors have attempted to establish dependencies between parameters that characterised sustainable resource recovery and valorisation.

For example, the study presented by Vea et al. concerns a review of life cycle assessment-based WMS-DSTs. Twenty-five WMS-DSTs were identified and analysed through a shortlisting procedure. Eight tools were shortlisted for the assessment of their applicability to deliver sustainability assessment of CBWMS. It was evaluated that only two tools consider both waste-specific heavy metals content in bioproducts and the associated implications when applied on soil. It was found that six tools model key properties that are necessary for assessing the environmental sustainability of CBWMSs, including waste-specific modelling emissions of gas, biogas generation or bioproduct composition.

Most of the shortlisted tools are flexible for simulating new technologies involved in CBWMS. In other studies (Ng et al., 2019) authors proposed a three-stage analysis of the sustainable resource recovery and valorisation system (Fig. 18.4): (1) multilevel system analysis; (2) scenario creation; and (3) sustainability assessment.

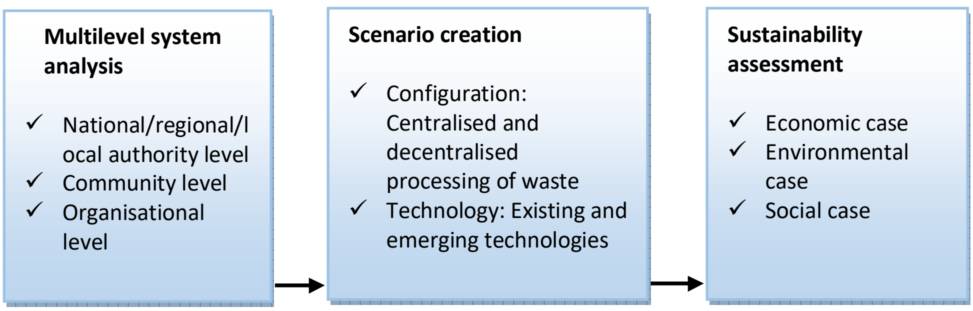


Fig. 18.4. Sustainable resource recovery and valorisation system (according to Ellen MacArthur Foundation, 2015)

Multilevel system analysis contains diagrams to examine the flow of resources from the national level, through the community level to the organisational level. Scenario creation contains configuration and technology of waste processing. The impacts and benefits have been examined through sustainability assessment which considers economic, environmental and social dimensions.

The waste streams have not been covered specifically by UE law, except for food waste, in terms of targets for separation and reduction. However, it is covered through the broad requirement to divert biodegradable waste from landfills and is therefore impacted by the relevant policies and requirements, such as landfill taxes. The European Environment Agency (2010) reported that although significant progress had been achieved in the recycling of technical nutrients, the same could not be said for biological nutrients. Nevertheless, this area has become a focus for research and development (R&D). The bioeconomy is a theme within the European Commission's Research and Innovation programme. The EU is providing R&D funding for the emerging EU bio-economy sector through the Bio-Based Industries (BBI) public private partnership. The BBI's funding calls for proposals aiming to accelerate 'the development of sustainable value chains from biomass feedstock supply via efficient processing, to the acceptance and application of bio-based products in the end-markets' (BBI, 2016). In other words, it will create realistic secondary markets and value chains, help reach critical mass and a high technology readiness level (TRL), de-risk investment, and pool resources. For the period between 2014-2020, the total number of Horizon 2020 funds allocated to this initiative is €3.7 billion, 25 percent (€975 million) being provided by Horizon 2020, and 75 percent (€2.7 billion) provided by industry. This includes €90 million of funding for calls for proposals related to biowaste valorisation included in two flagship calls for funding proposals (European Commission, 2016c).

18.6. CONCLUSIONS AND PERSPECTIVES

A successful biowaste management system needs good solutions in context of circular economy and sustainable requirements. Summarizing current data in the field of circular economy and its basic characteristics, bearing in mind specific properties of the considered type of waste, the following features demonstrating the circular nature of the given sector of economic activity can be proposed as below (Thomsen et al., 2017):

- Exclusion of waste disposal in landfills;
- Exclusion of pollutant emissions into the environment;
- Reuse/recycling/energy recovery out of the waste;
- Reduced input of primary natural resources, fossil fuels and electricity in comparison with the traditional model of the economy;
- Application of systems thinking, when at individual stages of a product life cycle, different enterprises, service providers or even allied industries are involved;
- Exclusion of accumulation of hazardous substances in the environment and living organisms.

The waste management sector shall become a crucial partner in new business models that focus on waste prevention and of course the waste management sector will have to "turn waste into resources" (European Commission, 2015). Maximising synergies between waste management and circular economy are the key to raising resource efficiency with co-benefits for job creation and economic prosperity.

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