Sanitary and environmental aspects of sewage sludge management

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1 Introduction

Globally, 80% of wastewater is discharged untreated into the world's waterways (Opec, 2018). Failure to treat effluents constitutes a serious threat to the environment, the climate, and human health, but this can also be considered a waste of resources. According to the goals of sustainable development and a circular economy, the wastewater shall be considered primarily as a source of water, then energy, organics, metals, and other resources. Consequently, sewage sludge is recognized as a source of renewable energy and material recovery (Christodoulou and Stamatelatou, 2016). Thus, no longer considered a "waste," it became a byproduct to be posttreated in order to be recycled into nature as energy, matter, or both. Sewage sludge is a reservoir of organic matter and nutrients, so it constitutes a potential substrate for a variety of possible reuse scenarios. Therefore, all potentially applied strategies shall fulfill the requirements of ecoinnovation. Thus, they shall lead to an important reduction of the negative environmental impacts by decreasing the consumption of natural resources or the release of harmful substances (Rorat and Kacprzak, 2017). Consequently, contemporary trends in organic waste treatment follow the sustainable development strategy in terms of environmental, economic, and social impacts. The composition of sewage sludge is highly variable and may depend on many various 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. For instance, extractable resources such as phosphorus (P) are predicted to become scarce or exhausted in the next 50–100 years, thus P recovery from wastewater is becoming an increasingly viable alternative (Connor et al., 2017). Nevertheless, a relatively low content of lignin and cellulose makes the organic matter easy to decompose. Hence, it degrades fast and can cause a sharp peak in the nitrate and pollutant concentration in the soil if applied without pretreatment. The biggest challenge, though, is seen in the contaminants, (1) organic (such as polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCB),

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adsorbable organohalogens (AOX), pesticides, surfactants, hormones, pharmaceuticals) and (2) inorganic (metals and their nanoparticles), and (3) pathogenic species of living organisms for example, bacteria, viruses, protozoa, and parasitic helminths (see review by Fijalkowski et al., 2017).

In this context, this chapter focuses on the sanitary and environmental dangers of the presence of the above-mentioned contaminants in sludge. The environmental risks of sludge spreading on soils will be presented as well as their possible treatment scenarios to propose an acceptable reuse of sewage sludge in a circular economy.

2 The global production of sewage sludge and the main directions of its management

At the European scale, the 91/271/ECC urban wastewater treatment directive adopted in May 1991 imposed the collection and treatment of wastewater in agglomerations with a population equivalent (PE) of more than 2000. Consequently, the substantial and constant increase of wastewater sludge and its disposal are becoming a growing challenge for municipalities in Europe. The annual sludge production in EU-27 will grow from 11.5 million tons of dry solids (DS) in 2010 to 13 million tons DS in 2020 (EC, 2008).

Table 1 shows the more recent data considering the production and disposal of sewage sludge for selected countries, according to OECD. While legislation more or less compels European countries to improve their sewage sludge management policies, many poor and developing countries are still studying possible wastewater treatment practices. For instance, the Federated States of Micronesia dumped almost 30% of produced sludge into the Pacific Ocean without any pretreatment in 2012 (Rouse, 2013).

The global population exceeded 7.5 billion this year and is expected to surpass 9 billion by 2050. Urban populations may rise nearly twice as fast as they are projected to nearly double from 3.4 billion in 2012 to 6.4 billion by 2050, especially in developing countries, where the number of people living in slums may rise even faster, from 1 billion to 1.4 billion in just a decade (Matiasi, 2012).

2.1 Sewage sludge as a substrate—Characteristics

The composition of sewage sludge is highly changeable during the process and also varies a lot between wastewater treatment facilities. Typically, raw (untreated) sewage sludge contains 2.0%–8.0% total dry solids (TS), 60%–80% of TS of volatile solids (VS), grease and fats, proteins, nitrogen, phosphorus, potassium, cellulose, iron, silica, alkalinity (mg/L as CaCO3), and organic acids (mg/L as Hac) (Metcalf and Eddy, 1991).

The potential danger of using raw sewage sludge (not stabilized, only mechanically treated) for the sewage-to-matter final disposal strategies is huge due to the presence of pathogenic organisms and other contaminants. Therefore, some stabilization processes shall be applied at the WWTPs. The choice of applied technology depends strongly on the characteristics of raw sludge. Some parameters are crucial for the processes of stabilization, for example, pH, organic acid content, and alkalinity limit the

Table 1 Sewage sludge production and disposal in selected countries in 2012

Country	Produced sewage sludge	Total disposal	Agricultural use	Compost and other applications	Landfill	Dumping at sea	Incineration
Austria	266	266	40	74	14	0	139
Belgium	157	107	19	n.d.	n.d.	0	89
Czech Republic	263	263	72	154	13	n.d.	8
Denmark	141	115	74	n.d.	1	0	34
Estonia	16	16	14	n.d.	2	0	..
Finland	141	141	7	93	10	0	32
France	987	932	684	n.d.	40	0	207
Germany	1849	1844	542	294	0	0	1009
Greece	119	119	14	0	40	0	39
Ireland	72	72	68	4	0	0	0
Israel	118350	n.d.	0	69311	3928	45111	0
Luxembourg	8	5	4	n.d.	0	0	1
Netherlands	346	325	0	0	0	0	321
Poland	533	533	115	33	47	0	57
Portugal	339	113	102	n.d.	11	0	0
Slovenia	26	26	0	2	1	0	13
Spain	2757	2577	1922	n.d.	384	0	100
Sweden	207	196	48	67	7	0	1
United Kingdom	1137	1078	844	n.d.	5	0	229

n.d., no data.
Data extracted on 13 May 2018, 16:08 UTC (GMT) from OECD.Stat.

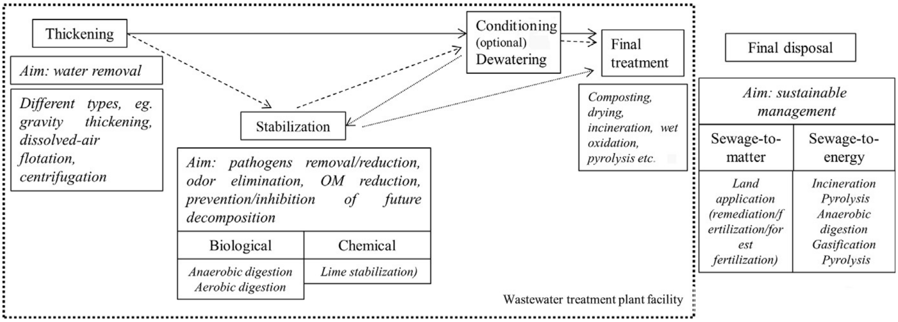

Fig. 1 Typical sewage sludge treatment process including processes at wastewater treatment plants (thickening/stabilization/dewatering) and the most common final disposal strategies applied worldwide.

anaerobic digestion process (Metcalf et al., 2013) while pH, ammonia, and other parameters are important for the composting and vermicomposting processes (Suleiman et al., 2017).

Fig. 1 presents the most typical pathways of sewage sludge treatment, including the processes at the WWTPs (thickening/stabilization/dewatering) and final disposal strategies. After thickening, sludge stabilization is usually performed. It is crucial for the further applications and aims primarily to reduce the potential risks by lowering the number of pathogens in organic matters. Two types of stabilization of liquid sewage sludge shall be distinguished:

1) Chemical stabilization by the addition of lime to alter the value of the pH to >11; eliminates the microbiological risk.
2) Biological stabilization, meaning anaerobic or aerobic digestion. Aerobic digestion is a process of treating the secondary sludge from the biological wastewater treatment process as activated sludge or trickling filters; anaerobic digestion can be conducted in low ($<10\%$) as well as medium (15%–20%) and high 22%–40% solid anaerobic digestion systems; this technology is described in detail in the next section.

After stabilization, the sludge shall be dewatered. Usually, this process is carried out using filter presses or centrifuges. As proper dewatering is crucial for the further disposal of sewage sludge, often an additional step of conditioning is required. The conditioners, synthetic organic polymers, or metal ions (typically iron salts) are used in order to coagulate the colloids in sludge and thus fasten the dewatering process (Novak, 2006). Only an efficient stage of water removal allows applying efficiently the selected techniques of final treatment and disposal of sewage sludge.

2.2 Sewage sludge final treatment and disposal

The main directions for sustainable sewage sludge management are:

a) Matter recovery (sewage-to-matter): use in agriculture (directly as a fertilizer) and remediation of devastated or degraded lands.
b) Energy recovery (sewage-to-energy) by incineration and alternate thermal methods as pyrolysis, quasipyrolysis and gasification or coincineration (in cement plants).

Wastewater contains a chemical energy that shall be converted to the usable form and thus fulfill a part of the worldwide need for renewable energy sources (Puyol et al., 2016). Different technologies can be used to transfer the excess sewage sludge into energy. This strategy has lately been of great interest. Primarily, it allows using the potential of sewage sludge without the environmental risks often discussed for land applications that could introduce contaminants into the soil. Moreover, it responds to a global call for renewable energy sources.

c) Others, such as landfilling or dumping at sea, which are forbidden by most countries but still practiced in some parts of the world, mostly in developing countries.

2.2.1 Anaerobic digestion

At first recognized mainly as a process of the stabilization of sewage sludge with the main aim of pathogen bacteria removal, now anaerobic digestion (AD) is often considered as a technically mature and cost-effective process that converts sludge into biogas (Cao and Pawłowski, 2012). The produced biogas can be used for the self-purposes of WWTPs that are characterized by a high demand for electricity, up to 0.78 kWh per m^3 of treated wastewater (Cano et al., 2015). Ideally, the process of anaerobic digestion shall fulfill this high demand. In order to increase efficiency and thus biogas yields, it was proposed to introduce the other ingredients in the so-called codigestion. For instance, Grosser (2018) used grease trap sludge and an organic fraction of municipal waste as cosubstrates in the process, which enhanced the efficiency of sewage sludge anaerobic digestion. Similarly, other organic wastes have been tested with success, for example, food waste, cheese whey, and olive mill wastewater (Maragkaki et al., 2018). Thus, codigestion of sewage sludge can be understood as a method of management of different organic wastes. Nevertheless, it cannot be fully seen as a final disposal of sludge, as it generates another byproduct, digested sludge (digestate), that still contains a high quantity of nutrients and contaminants and must be treated. It was proven that the utilization of digestates may replace or reduce the use of mineral fertilizer in agronomic plant production, as it is rich in plant-available nutrients (ammonium, phosphate, and potassium) (Sogn et al., 2018). Yet, in-land use as a biofertilizer is possible only if the product can be qualified according to applicable norms, usually regulated by soil protection legislation, fertilizer, or waste legislation. Otherwise, other options shall be considered. Lately, Peng et al. (2018) proposed using the digestate in landfill bioreactors in order to remove the nitrogen of old landfill leachate. Digestate was also successfully applied with other organic wastes as the organic fraction of municipal wastes, sawdust, and green wastes in the process of vermicomposting (Rorat et al., 2017). As the chemical composition of digestate corresponds to the composition of the used substrates, the long-term effects of its introduction to the soil shall be studied in order to appreciate the impact on soil functions (soil biodiversity and microbial cycles). The existing studies focus mainly on the fertilizer properties of the digestates produced from different substrates. According to Nkoa (2014), the most common risks associated with the application of digestate in land are related to:

a) Risks of atmospheric pollution (ammonia emission and fallout, nitrous oxide emission).
b) Risks of nutrient pollution (excess nitrogen and phosphorus).
c) Risk of soil contamination (chemical/biological contamination).

2.2.2 Composting and vermicomposting

Composting as a method of biological decomposition of biowastes in the presence of oxygen contributes powerfully to the recycling and conservation of several macro- and micronutrients from sewage sludge in the soil. Its alternative, vermicomposting, is a modern, inexpensive, and eco-friendly biotechnology in which earthworms are employed as natural bioreactors in order to decompose the organic matter (Suleiman et al., 2017). Their metabolic activity and cooperation with microorganisms lead to a 40%–60% reduction of volume, an increase of bioavailability of nutrients to plants, a reduction in the C/N ratio, and a decrease of the availability of some dangerous contaminants such as metals (Rorat et al., 2015). Although composting can be considered highly beneficial and a low cost sewage-to-matter strategy that allows recycling organic nutrients into the ecosystem, it still causes some important problems from an environmental point of view. Due to the rapid degradation of nitrogenous organic matter, important nitrogen losses and greenhouse gas (GHG) emissions can be noted (Sánchez-Monedero et al., 2010). Those effects can be partially reduced by the introduction of different bulking agents, for example, agricultural wastes and alkaline amendments such as lime, zeolite, and bentonite. Recently, biochar has also been considered an efficient agent causing a reduction of greenhouse gases, ammonia, and extractable ammonia emissions (Malińska et al., 2014; Awasthi et al., 2016). Although these effects can be partially eliminated, researchers are concerned about the input of potentially toxic metal elements and therefore their possible accumulation for several in the soil horizon (Fang et al., 2017). The same type of risk is related to the presence of other chemical compounds and pharmaceuticals as well as some pathogens that can survive the process.

2.2.3 Thermal processes

All thermal processes are considered sludge-to-energy systems that lead to the complete oxidation of the volatile matter with production of a residue (ash). Generally, the most famous technologies are incineration, gasification, pyrolysis, and plasma gasification. Combustion and/or incineration are considered the most attractive disposal methods for sewage sludge in Europe (EC, 2008), as they replace potentially dangerous landfilling and agricultural strategies. This also allows largely reducing the volume of sewage sludge to destroy the microbiological danger, minimize the odors, and simultaneously recover the renewable energy. Three main variants of this process are used: incineration in dedicated plants, coincineration with municipal solid wastes, and incineration in cement kilns. The environmental cost related to those systems is mostly related to the high energy consumption and production of harmful gaseous emissions (i.e., dioxins and furans) (Garrido-Baserba et al., 2015). Moreover, the ashes coming from the process can be considered a concentrated pollutant that accumulates chemical contaminants. The interesting alternative for ash disposal is a cement replacement. After incineration, sewage sludge is still rich in silica, alumina, calcium oxide, and iron oxide, so it can be used in the production of building materials. Moreover, in this form, metals are stabilized and solidified, so the potential risk is reduced (Samolada and Zabaniotou, 2014).

Pyrolysis is being recognized as a relatively expensive but highly effective technology. Basically, the process converts the organic matter into bioenergy (oil/gas) with a byproduct in the form of so-called biochar. Thus, pyrolysis allows yielding a major bio-oil fraction potentially useful as a fuel or as a source of chemical products. Generally, pyrolysis and similar processes of combustion of sewage sludge are regarded as endothermic. Nevertheless, Atienza-Martínez et al. (2018) have shown lately that the necessity of pretreatment of this substrate (dehydration) moves it more to the exothermic processes, although it is still the most cost-consuming throughout the scenario. Thus, improvement of the steps allowing water removal is crucial for the future potential of the process. Independently, numerous studies have shown many advantages of using biochar for environmental management. For instance, it can be applied for soil improvement, to improve the efficiency of the resources, for remediation and/or protection of lands, and future greenhouse gas mitigation (Joseph and Lehmann, 2015). In general, biochar can thus be defined as a solid, carbon-rich material obtained in the process of zero or low oxygen pyrolysis from different C-based feedstocks, which, applied to the soils, sustainably sequesters carbon and thus improves soil quality in the long term (Verheijen et al., 2010).

As far as the process of combustion eliminates the microbiological risk related to the land application of sewage sludge, still some questions considering chemical pollutants are being posed. These need to be evaluated on a case-by-case basis, not only with concern to the biochar product itself but also to soil type and environmental conditions (Verheijen et al., 2010). Nevertheless, lately it has been recognized that the addition of activated carbon or biochar to sewage sludge immobilizes the bioavailable fractions of polycyclic aromatic hydrocarbons and metal elements (Kończak and Oleszczuk, 2018). Moreover, Frišták and Soja (2015) recognized that the addition of biochar produced from wood chips and garden residues into the sewage sludge and its application as soil amendments have increased the content of available forms of phosphorus. The positive effects of the addition of sewage sludge-derived biochar were also observed during the process of vermicomposting of sewage sludge, where it significantly reduced the bioavailability of Cd and Zn for *Eisenia fetida* earthworms (Malińska et al., 2017).

3 Sewage sludge as sources and drive pathways for contaminants

Considering all valuable resources present in sewage sludge (organic matter, plant culture available nutrients), many countries recognized this byproduct as a potential substrate for fertilization in agriculture or remediation of polluted areas. Nevertheless, sewage sludge applications on agricultural land might contribute to the dispersal of a broad range of unwanted constituents on soils possibly used for food production. Such undesirable contaminants (potentially toxic elements (PTE) such as metals, trace organic compounds (TrOC), and pathogenic organisms) may pose sanitary and environmental risks (Andreoli et al., 2017). Toxic pollutants in sewage sludge could even, in some cases, increase preexisting environmental problems and lead to secondary environmental contaminants and poisonings if not properly managed.

As a reflection of our chemical-based consumer society, our wastewater and sewage sludge mirror compounds we use, produce, release, and discharge. Sludge composition, its agronomic interests, and contaminations may so differ greatly. Such parameters depend on the wastewater origins (agricultural, industrial areas or urban), the local household and consumer habits, the sewer collection (separation or not for wastewater and runoff), the regional environmental regulations, the season and obviously of the size and the process used by the considered WWTP.

3.1 Chemical contaminants in sludge

The environmental risk of sludge contaminants and their concentrations in soil after land application is dependent on their initial concentrations (in both soils and sludge) and the application rate (cumulative effects), management practices, and losses. Therefore, volatile and easily degradable contaminants may still pose environmental risks in the case of high initial concentrations and repeated applications (Harrison et al., 2006).

There are two environmental and public health issues involved with the use or disposal of WWTP biosolids: potentially toxic elements (PTEs) and organic contaminants (OCs), chiefly persistent chemicals. Concerns relate to potential trophic transfers (via cultivated plants) and possible contamination of groundwater. PTE is a common general term that includes metal elements, formerly known as "heavy metals" or "metal trace elements." They are naturally present in soils and the current concerns come more from anthropogenic soil contamination by these elements through the use of fertilizers (including sludge, slurries, and manures), pesticides, and poor waste management. They classically comprise the following metals and metalloids: arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), molybdenum (Mo), nickel (Ni), and zinc (Zn). Potential environmental risks associated with these PTEs in the context of sludge land applications have been extensively studied and environmental guidelines and regulations defined. For several OCs, such a regulatory framework exists as for polycyclic aromatic hydrocarbons and several persistent organic pollutants (POPs, such as PCBs, dioxins, and related compounds (PCDD/F)). The list and limit values for concentrations of metal elements and OCs that should restrict the use of sewage sludge in agriculture have been updated recently in Europe (EC, 2000). They have suggested the regulation of linear alkylbenzene sulfonates (LAS; extensively used as surfactants), Di(2-ethylhexyl) phthalate (DEHP), nonylphenol and nonylphenol ethoxylates (NP(E)), halogenated organic compounds (i.e., adsorbable organic halides (AOX)), PAH, PCBs, and polychlorinated dibenzo-p-dioxins and dibenzo-furans (PCDD/F). Studies on sludge contaminants started in the 1970s. Most of the chemicals we use in our everyday lives and those with industrial applications are likely to end their lifecycle in biosolids, not to mention byproducts of human activities. We could only exclude highly volatile products and those that are rapidly degraded. In addition to the OCs in urban wastewaters, surface run-off of atmospheric-deposited environmental contaminants onto artificialized areas (concreted and paved, consequently impervious) contribute to the accumulation in sewage sludge of lipophilic compounds that tend to adsorb to solid particles. This situation is like that of sediments in aquatic ecosystems. It is the nonpolar and some persistent compounds that could represent an environmental risk with sludge recycling. One of the most concrete examples may be that of surfactants such as LAS (little worrying, even considering present uses but above all realized risk assessments, (Schowanek et al., 2007)) and more worrisome fluorosurfactants such as perfluorooctanesulfonic acid (PFOS), perfluorooctanoic acid (PFOA), and perfluorononanoic acid (PFNA) (Smith, 2009).

As an introduction to their review of emerging OC in sludge, Clarke and Smith (2011) considered that of the 100,000 chemicals registered in the EU, all are likely to be found in WWTP sludge. In 1996, Wilson et al. (1996) proposed a list of 300 products to prioritize in sludge research work and surveys (priority organic pollutants on environmental agency and country lists and compounds identified in sludge worldwide). This number is close to the one (332) from (Drescher-Kaden et al., 1992) on toxic or suspected toxic OC residues in German sludge, based on a review of the literature on more than 900 articles. Harrison et al. (2006) reported 516 OCs for which concentration data were available in the peer-reviewed literature and official government reports. More recently, Eriksson et al. (2008) concluded after a literature review that 541 OCs could potentially be present in sewage sludge due to their use in construction materials, pharmaceuticals, and personal care products. However, some OC concentrations are not sufficiently characterized in such matrices as sludge and biosolids because their analyses are confined on regulated or identified contaminants on priority lists that represent only a small fraction of the present OCs (Harrison et al., 2006). These authors highlight the lack of knowledge for OCs as nitrosamines with high environmental risks while the knowledge is more comprehensive for some families such as pesticides, PAHs, and PCBs. Thus, a global vision of the contaminants present in the sludge is possible for some as metals, PCB, PAH ... Besides, the concentrations for these compounds in sludge tend to decrease. As an illustration of sludge contaminations around the world, we propose Tables 2–3. Such an overall worldwide inventory could also be made for PCBs and PCDD/Fs, but it would not be possible for most other TrOCs.

The first lesson of Table 2 on metal concentrations in sludge is that there is a real scientific consensus for these elements. Zn is always the predominant metal in terms of concentrations. PTE is one of the few families of contaminants for which sequential or selective extraction approaches could be used to approach the key issue in contaminant risk assessment, which is bioavailability. The origins of PTEs in wastewater are also well described and understood and even some technical solutions for metal removal from sludge have been developed and proposed (see (Babel and Del Mundo Dacera, 2006) for an example of review).

The fact that PAHs (Table 3) are priority environmental pollutants relies mainly on their possible harmful effects on biota as well as carcinogenicity in humans. They are too lipophilic with low biodegradability and they accumulate in sludge, sediments, and soils. The main sources of sludge PAH are industrial wastes as well as domestic sewage, atmospheric rainfall, precipitation of airborne pollutants, and road surface and tire abrasion products (PAHs) (Bomboi and Hernandez, 1991). We will emphasize the study of (Stefaniuk et al., 2018), which proposes to analyze the freely

Table 2 Total concentrations (averages of selected metals present in sewage sludge of different countries worldwide)

	Total element concentrations (mg/kg of dry sludge solids; except specific units for several values)											
	Ag	Al	As	Cd	Cr	Cu	Fe	Hg	Ni	Pb	Zn	Comments and references
Brazil	n.r.	n.r.	14.69	10.75	143.72	255.39	n.r.	2.35	41.99	80.37	688.83	Averaged values in wastewater sludge (UN-HABITAT, 2008)
Canada	n.r.	n.r.	n.r.	2.3–10	66–2021	180–2300	n.r.	n.r.	37–179	26–465	354–640	(Tyagi et al., 1988; Benmoussa et al., 1997; Filali-Meknassi et al., 2000) In (Pathak et al., 2009)
	n.r.	n.r.	1	1	50	460	n.r.	1	16	51	593	Averaged values in Biosolids, Ottawa; British Columbia and Greater Moncton Sewerage Commission (UN-HABITAT, 2008)
	n.r.	n.r.	4.6	2.3	50.7	888	n.r.	3.1	26.4	56	588	
	n.r.	n.r	n.r.	0.5	n.r.	137	n.r.	0.3	9	27	223	
China	n.r.	n.r.	16.7–26	5.9–13	45.8–78.4	131.2–394.5	n.r.	17–24	49.3–95.5	57.5–109.3	783.4–3096	Min and max six plants (Dai et al., 2007)
	n.r.	n.r.	n.r.	2.1–19.4	22.2–453.2	210.6–1191.3	n.r.	n.r.	25.1–106.6	41.2–452.2	1406.2–3699.2	Min and max 12 plants in Zhejiang Province (Hua et al., 2008)
	n.r.	n.r.	n.r.	1.65	1983.8	3323.9	n.r	n.r.	422.0	69.7	2424.2	Xiamen WWTP, Fujian Province (Wang et al., 2016)
	n.r.	n.r.	n.r.	11.72, 9.63–15.13	112.5, 50–125	383.47, 62.75–796.63	n.r..	n.r.	692.94, 98.63–2180.13	113.19, 86.25–136.75	609.44, 290.38–831	Mean, min and max four plants (Yang et al., 2017)
	n.r	n.r.	n.r	2.12	514.24	266.06	n.r	n.r	84.64	n.r.	1345.51	Raw sludge from the Shihezi WWTP, Xinjiang (Li et al., 2018)
Colombia	n.r.	n.r.	18.6	76	72.5	163.4	n.r.	8	42.9	87.5	1014.2	Averaged values in biosolids from El Salitre WWTP, Bogota (UN-HABITAT, 2008)
Finland	n.r.	n.r.	n.r.	0.6	18.1	244	n.r.	0.37	30.3	8.8	332	Averaged values in 2005 wastewater sludge (UN-HABITAT, 2008)
France	n.r.	n.r.	n.r.	4.5	64	286	n.r.	2.1	35	107	761	Median of 237 mainly domestic WWTP, (ADEME, 1995)
Germany	n.r.	n.r.	n.r.	1.5	50	275	n.r.	n.r.	23.3	67.7	834	(CEC, 1999). In (Pathak et al., 2009)
	n.r.	n.r.	n.r.	1.52	60.5	380.2	n.r.	0.92	32.2	61.7	955.7	Averaged values in 2003 wastewater sludge—DWA survey (UN-HABITAT, 2008)

Hong-Kong	n.r.	n.r.	n.r.	n.r.	663	112–255	n.r.	n.r.	44.5–622	52.5–57	1009–2823	(Xiang et al., 2000; Wong and Selvam, 2006) In (Pathak et al., 2009)
India	n.r.	n.r.	n.r.	41–54	102–8810	280–543	n.r.	n.r.	192–293	91–129	870–1510	(Singh et al., 2004; Pathak et al., 2008) In (Pathak et al., 2009)
Iran	n.r.	60–259	n.r.	6.1–15.3	2782–8071	57.5–163	n.r.	n.r.	17.9–59.3	n.r.	260–2077	Mean, min and max four plants (Feizi et al., 2018)
Ireland	n.r.	n.r.	n.d.	12	35	520	n.r.	n.d.	18	252	886	Averaged values from 16 plants (Healy et al., 2016)
Italy	n.r.	n.r.	7.3	2.7	62.1	601	n.r.	n.r.	29.2	16.3	961	Sludge from the municipal WWTP of Brindisi (Nissim et al., 2018)
	n.r.	n.r.	n.r	2.1	n.r.	370	n.r.	n.r.	19	72	1500	(Lazzari et al., 2000) In (Pathak et al., 2009)
	n.r.	n.r.	n.r	0.3–0.9	18–65	90–206	n.r.	0.2–0.9	11–15	80–126	283–895	Range three plants (Gianico et al., 2013)
	n.r.	n.r.	n.r	1.6	22.3	261	n.r.	0.2	15.6	76.2	577	Averaged values in 2006 Sardinia biosolids used in agriculture (UN-HABITAT, 2008)
Japan	n.r.	n.r.	8.2	2.2	19.5	n.r.	n.r.	1.1	32.3	5.2	n.r.	Averaged values in dried wastewater sludge, Suzu (UN-HABITAT, 2008)
Poland	n.r.	n.r.	n.r.	n.d.	58	194	n.r.	1.04	22	23.5	1459	Mean values in sludge from the treatment plant of Sokółka (Wołejko et al., 2018)
	n.r. n.r. n.r.	n.r. n.r. n.r.	n.r. n.r. n.r.	1.63 n.d. 1.36	16.1 17.2 37.2	191.4 158.0 55.8	n.r. n.r. n.r.	0.457 0.297 0.234	15.6 17.6 n.d.	32.4 50.4 n.d.	1248.5 962.9 344.8	Sludge from a small, a medium and a large WWTP in Central Poland (Urbaniak et al., 2017)
Netherlands	n.r.	n.r.	**7.3**; *n.d.*	**3.8;** *n.d.*	**66**; *20–40*	**420**; *10–100*	n.r.	**1.9**; *n.d.*	**36**; *7–22*	**210**; *2–12*	**1100**; *60–400*	Range values from Dutch **municipal** and *agro-industrial* WWTP sludge (Veeken and Hamelers, 1999)
Russia	n.r.	n.r.	n.r.	n.r.	305–310	200–300	n.r.	11.35	75–77	34.7	0.07–0.08 (%)	(Nikovski and Kalinichenko, 2014) In (Fijalkowski et al., 2017)
	n.r.	n.r.	0–24	0–300	18.2–1280	0.9–1200	n.r.	0–11.35	1.4–306	0.8–1070	3.0–3820	Range values in 2017 sludge from Moscow Area (UN-HABITAT, 2008)
Slovenia	n.r.	n.r.	2	1	90	200	n.r.	2	35	150	600	Averaged values in wastewater sludge (UN-HABITAT, 2008)

Continued

Table 2 Continued

	Total element concentrations (mg/kg of dry sludge solids; except specific units for several values)											Comments and references
	Ag	Al	As	Cd	Cr	Cu	Fe	Hg	Ni	Pb	Zn	
Spain	n.r.	n.r.	n.r.	2.37–18.3	54.4–3809	204–337	n.r.	n.r.	23.2–36.5	167–223	871–1626	(Alvarez et al., 2002) In (Pathak et al., 2009)
Sweden	1.98	n.r.	n.r.	2.10	n.r.	323	n.r.	1.45	17.3	45	720	Mean, min and max 11 plants (Östman et al., 2017)
	0.72–3.26			0.59–37		110–640		0.19–10	7.3–36	10.9–560	396–1500	
South Africa	n.r.	n.r.	n.r.	0.82–3.10	35.07–134.08	263.68–626.00	n.r.	n.r.	31.34–51.43	21.28–171.87	1031.75–1732.00	Min and max 5 plants, Limpopo province (Shamuyarira and Gumbo, 2014)
Turkey	n.r.	n.r.	n.r.	1.24	34.2	70.2	n.r.	n.r.	62.1	34.2	300	Averaged values in Izmir et al., 2008) Guneybati WWTP sludge (UN-HABITAT, 2008)
UE countries	0.1–14.7	0.1–60 (%)	5.1–56.1	0.3–5.1	10.8–1542.2	27.3–578.1	0.2–14.9 (%)	0.1–1.1	8.6–310	4.0–429.8	0–0.1 (%)	Range values adapted from (Gawlik, 2012) In (Fijalkowski et al., 2017)
	n.r.	n.r.	n.r.	0.4–3.8	16–275	39–641	n.r.	0.3–3	9–90	13–221	142–2000	Typical ranges from (ICON, 2001) In (Gianico et al., 2013)
UK	n.r.	n.r.	n.r.	3.5	159.5	562	n.r.	n.r.	58.5	221.5	778	(CEC, 1999; EU, 1999) In (Pathak et al., 2009)
USA	n.r.	n.r.	n.r.	25	178	616	n.r.	n.r.	71	170	1285	(Bastian, 1997) In (Pathak et al., 2009)
	n.r	n.r.	2.6	2	n.r	670	n.r.	1.3	16	39	743	Averaged values in class B biosolids—Denver and Los Angeles Hyperion Treatment Plant (UN-HABITAT, 2008)
	n.r	n.r.	6.06	10.2	84	1060	n.r.	1.91	50.8	38.5	1180	

WWTP, wastewater treatment plant; *n.r.*, not reported; *n.d.*, not detected or under quantification limits.

dissolved PAH concentrations rather than the total concentrations in order to better estimate their potential environmental availabilities.

For so-called "emerging" contaminants, scientific works appear and a global scheme is emerging since a decade. These contaminants are considered emerging because either the current analytical techniques finally allow their analyses in sludge, or the new industrial and domestic uses of certain products increase their concentrations in environmental matrices. Among these compounds are prominent pharmaceuticals, personal care products and residues, endocrine disruptors, and, more recently, nanoparticles and microplastics. In the excellent review of Clarke and Smith (2011), these authors ranked the following biosolid emerging contaminants (priority decreasing order): PFOS and PFOA, polychlorinated alkanes, polychlorinated naphthalenes, organotins, polybrominated diphenyl ethers, triclosan, triclocarban, benzothiazoles, antibiotics and pharmaceuticals, synthetic musks, bisphenol A, quaternary ammonium compounds, steroids, phthalate acid esters, and polydimethylsiloxanes.

The field of such OCs in sludge as well as their fates and behaviors following sludge land application are largely to be investigated to finally allow their comprehensive risk assessment and case-by-case sludge environmental (even ecotoxicological) assessments must be favored.

3.2 Pathogenic organisms

Sewage sludge contains biological agents that can be problematic for living organisms because some are pathogenic or may simply disturb natural ecosystems. Generally, four groups of pathogens can be found in sewage sludge: viruses, bacteria, parasites, and fungi. In fact, due to its very rich organic matter, sewage sludge can include many bacteria and fungi species in large quantities (Fijalkowski et al., 2017). Other organisms such as viruses and parasites are also regularly present in sewage sludge (Frąc et al., 2014). The concentration and type of pathogen depend on the type of WWTP, the source of wastewater, and some environmental factors (Romdhana et al., 2009). However, the majority of these pathogenic organisms are derived from human or animal feces (Bloem et al., 2017).

The microbial flora present in sewage sludge is very diverse and abundant due to the high content of organic matter. The majority of these bacteria are saprophytes; they are safe and play an important role in the process of wastewater treatment by forming flocs and degrading some contaminants (Tozzoli et al., 2017). However, some of these bacteria are pathogenic. Huang et al. (2018) identified 243 potentially pathogenic bacterial species in activated sludge, including six major pathogens (*Bacillus anthracis*, *Clostridium perfringens*, *Enterococcus faecalis*, *Escherichia coli*, *Pseudomonas aeruginosa*, and *Vibrio cholera*) that can reach abundances of 14% of the bacterial flora. Others pathogens such as *Salmonella*, *Shigella*, *Klebsiella*, *Serratia*, *Enterobacter,* or *Proteus* have also been identified (Korzeniewska, 2011). All these bacteria may cause various infections such as urinary tract infections (*E. coli*), pneumonia (*Klebsiella* and *Enterobacter*), blood infections (*Enterobacteriaceae*), and gastrointestinal infections (*E. coli*, *Salmonella*). These diseases can appear after contamination by gastrointestinal, respiratory, urinary, and biliary tracts (Korzeniewska, 2011).

Table 3 Total concentrations (average or range values) of selected PAH congeners in sewage sludge of different countries worldwide

	Concentrations of individual PAH (μg/kg of dry sludge solids except specific units for several values)								
	Na	**Ace**	**Ac**	**Fl**	**Phen**	**Ant**	**Fluo**	**Pyr**	**B[a]A**
China	16.23–180.95	20.08–289.86	3.11–131.14	17.37–91.77	48.37–466.41	22.46–214.44	138.40–658.33	120.47–317.31	170.58–2171.24
	140–16320	20–6570	**0**–3890	490–11940	**0–35100**	40–6140	620–9880	**0**–15680	180–4820
Czech Republic	n.r.	n.r.	n.r.	n.r.	17–3910	n.r.	12–877.9	9.5–2869.9	n.r.
France	n.r.	n.r.	n.r.	*n.r.*	n.r.	n.r.	40 **1930** **1070**	n.r.	n.r.
Italy	n.r.	0	0	0	0	0	0	0	0
	n.r.	n.r.	**1–184**	1–228	1–673	n.r.	2–844	1–1118	n.r.
Jordan	1.9 3.0 1.1	n.r.	**0.1** 0.3 0.4	0.5 1.9 3.5	3.0 4.8 **7.7**	0.4 3.9 0.5	**4.8** 3.3 1.4	4.2 6.2 4.8	2.0 **0** 0.7
Japan	n.r.	3.3	2.5	3.9	12.6	2.6	14.7	**15**	3.6
Malaysia	**4550–5380**	n.r.	n.r.	n.r.	**0**–4070	n.r.	**0–0**	n.r.	n.r.
Poland	**0**	**0**	**0**	**0**	68.7	17.9	**398.2**	**0**	84.8

Ch	B[b]F	B[k]F	B[a]P	D[ah]A	B[ghi]P	Ind	∑PAH* (μg/kg dry matter)	Comments and references
493.89–2958.66	572.71–**8514.09**	115.07–2138.07	226.73–6174.17	**0**–1134.62	**0**–1038.67	**0**–2682.70	2467.32–259723.79	Min and max six plants, Beijing *(16 US EPA) (Dai et al., 2007)
470–11200	**0**–1080	0–2170	230–7850	**0**–14050	**0**–6520	**0**–9370	33730–87500 13890–641200	Min and max 12 plants *(16 US EPA and 9 EC) (Hua et al., 2008)
n.r.	**434.4–3802.9§**	**434.4–3802.9§**	21.5–2468.4	n.r.	312.1–2724.6	305.3–2905.4	1481.3–17313.6	Min and max values of 45 samples of sludge (Vácha et al., 2005) In (Suciu et al., 2015)
n.r.	30 630 240	n.r.	**90** **800** 370	n.r.	n.r.	n.r.	n.r.	Min, max and mean in sludge collected along the treatment process of Seine Aval treatment plant (Blanchard et al., 2004)
0	0	0	0	0	0	0	n.r.	Sludge from the municipal WWTP of Brindisi (Nissim et al., 2018)
n.r.	2–**1511§**	**2–1511§**	1–1341	n.r.	1–1030	1–1310	11–3917	Min and max values from 35 WWTP in northern Italy 452 samples—survey of four years *(9 EC) (Suciu et al., 2015)
1.9 1.8 1.2	2.1 0.9 **0**	1.0 0.4 0.3	1.9 1.2 0.5	0.4 **0** **0**	4.7 **7.3** 6.3	1.4 1.2 **0**	34.6 39.3 28.7	WWTP sludge from an University complex or a municipal area and sludge from a raw wastewater and sludge disposal site, Karak *(∑16 with B[e]P instead of Ace) (Jiries et al., 2000)
3.6	4.4	3	4.3	1.6	2.8	**1.6**	82	Mix of six dewatered WWTP sludge: rural, urban and residential areas *(∑16 with B[e]P instead of Na) (Ozaki et al., 2017)
n.r.	n.r.	n.r.	**0–0**	n.r.	n.r.	n.r.	n.r.	Min and max from three plants, Johore (Ahmad et al., 2004)
0	214.5	**0**	134.7	**0**	**0**	88.2	2039.9	Min, max and mean of 15 municipal sewage treatment plants *(16 US EPA) (Baran and Oleszczuk, 2003)

Continued

Table 3 Continued

	Concentrations of individual PAH (*μ*g/kg of dry sludge solids except specific units for several values)								
	Na	**Ace**	**Ac**	**Fl**	**Phen**	**Ant**	**Fluo**	**Pyr**	**B[a]A**
	0 **0** **0**	**7105.1** **2132.4** 0.0079	2946.5 886.6 0.0885	974.2 199.6 0.096	1149.2 526.7 0.1299	425.5 165.9 0.2822	5399.9 1937.8 **0.6060**	5050.5 1096.3 0.6115	2579.3 918.5 0.1295
	80.5	2.03	32.1	25.4	7.81	4.88	17.1	11.3	0.47
Portugal	**(ng/L)** 27–198 **0**–309	(ng/L) **0**–118 **0**–30	(ng/L) **0**–492 **0**–72	(ng/L) 28–704 77–909	(ng/L) 540–**2030** 250–**1760**	(ng/L) 34–234 **0**–292	(ng/L) 100–629 56–685	(ng/L) 277–702 112–706	(ng/L) 80–184 29–155
Romania	**0.575** **0.409**	n.r.	0.004 0.007	0.022 0.026	0.070 0.060	0.005 0.004	n.r.	0.024 0.000	0.009 0.004
Spain	n.r.	n.r.	**0**–300	**0**–750	20–**3000**	n.r.	33–570	**58**–731	n.r.
Turkey	n.r.	n.r.	n.r.	n.r.	**0**–1164.5	n.r.	12–**877.9**	9.5–2869.9	n.r.
UE countries	n.r.	n.r.	n.r.	n.r.	29.9–552.2	15.3–724.0	34.5–**3126.8**	47.2–2637.0	**9.1**–1832.6
	 n.r.	 n.r.	 n.r.	 n.r.	100% 29.9–552	84% n.r.	100% 34.5–3217	100% 47.2–2637	97% n.r.
UK	n.r.	n.r.	1700–6600	**3600**–8100	3200–**16000**	n.r.	1400–7400	2100–5600	n.r.

Less and more present congener reported values in a table line are in bold. Abbreviations: *n.r.*, not reported; *0*, not detected or under detection or quantification limits. *Na*, Naphthalene; *Ace*, Acenaphthylene; *Ac*, Acenaphtene; *Fl*, Fluorene; *Phen*, Phenanthrene; *Ant*, Anthracene; *Fluo*, Fluoranthene; *Pyr*, Pyrene; *B[a]A*, Benz[a]anthracene; *Ch*, Chrysene; *B[b]F*, Benzo[b] fluoranthene; *B[k]F*, Benzo[k] fluoranthene; *B[a]P*, Benzo[a]pyrene; *B[e]P*, Benzo[e]pyrene; *D[ah]A*, Dibenz[a,h] anthracene; *B[ghi]P*, Benzo[ghi]perylene; *Ind*, Indeno[1,2,3-cd]pyrene.

Ch	B[b]F	B[k]F	B[a]P	D[ah]A	B[ghi]P	Ind	∑PAH* (μg/kg dry matter)	Comments and references
1869.1	7572.5	**0**	1786.0	458.9	835.1	1395.6	36034.1	
577.3	1857.3	**0**	610.7	72.3	262.6	368.8	11612.9	
0.1543	0.0715	0.0524	0.0423	0.0822	0.0176	0.1295	n.r.	Mean values in sludge from the treatment plant of Sokółka (Wołejko et al., 2018)
0.85	0.12	0.13	0.12	**0.012**	0.022	0.028	183.1	Freely dissolved PAHs in municipal WWTP sludge, Chełm *(16 US EPA) (Stefaniuk et al., 2018)
(ng/L)	(ng/L)	(ng/L)	(ng/L)	**(ng/L)**	(ng/L)	(ng/L)	(ng/L)	
79–312	35–479	11–289	23–522	**0**–66	**0**–589	**0**–461	2510–5520	Range values from two domestic and four industrial WWTP sludge *(16 US EPA) (Pérez et al., 2001)
13–283	**0**–234	**0**–95	17–275	**0**–125	0–160	27–295	1130–4120	
0	0.005	0.001	0.003	0.009	0.032	0.022	n.r.	Averaged values in primary and in digested dehydrated sludge of Cluj-Napoca WWTP (Alhafez et al., 2012)
0	0.002	**0**	0.001	0.003	0.023	0		
n.r.	**0**–242§	**0**–242§	17–100	n.r.	**0–60**	**0**–88	308–5118	Min and max values of 38 samples of sludge * (9 EC) (Sánchez-Brunete et al., 2007) In (Suciu et al., 2015)
n.r.	434.4–**3802.9§**	**434.4–3802.9§**	21.5–2468.4	n.r.	312.1–2724.6	305.3–2905.4	1481.3–17313.6	Min and max values of 14 samples of sludge * (9 EC) (Salihoglu et al., 2010) In (Suciu et al., 2015)
21.0–2020.5	25.1–1919.4	9.9–1048.0	17.9–1475.5	n.r.	n.r.	n.r.	*n.r.*	Range values and occurrence adapted from (Gawlik, 2012) In (Fijalkowski et al., 2017)
94%	91%	100%	100%					
n.r.	**9.9–2967§§**	**9.9–2967§§**	17.9–1476	*n.r.*	29.7–1335	24.2–1401	*n.r.*	Min and max values of 32 samples of sludge from member states (JRC, 2012) In (Suciu et al., 2015)
n.r.	1800–11700§	1800–11700§	690–4000	n.r.	470–**2300**	**390**–2700	18000–50000	Min and max values of 14 samples of sludge * (9 EC) (Stevens et al., 2003) In (Suciu et al., 2015)

16 US EPA priority PAH: acenaphtene, acenaphthylene, anthracene, benzo[a]anthracene, benzo[b]fluoranthene, benzo[k] fluoranthene, benzo[ghi]perylene, benzo[a]pyrene, chrysene, dibenzo[a,h]anthracene, fluoranthene, fluorene, indeno[1,2,3-cd] pyrene, naphthalene, phenanthrene and pyrene.
9 CEC: EU proposed sum of PAHs (acenaphthene, fluorene, phenanthrene, fluoranthene, pyrene, benzo[b + j + k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene, benzo[ghi]perylene) should not exceed 6 mg/kg dry matter. in sludge for land application) (EC, 2000) ; § reported concentrations for benzo[b + j + k]fluoranthene; §§ reported concentrations for benzo[b + k] fluoranthene.

E. coli is already part of one of the quality criteria for sludge (European Directive 86/278/EEC) (EC, 1986). Also, *Salmonella* is one of the most-studied bacteria in WWTP sludge (Jr Krzyzanowski et al., 2016). These bacteria can survive once released into the environment in part through sludge spreading on agricultural plots (Jr Krzyzanowski et al., 2016; Bloem et al., 2017; Ellis et al., 2018). Thus, the consumption of food from these lands could be a way of contamination. It has been shown that even with low concentrations of *Salmonella* in sludge, some vegetables such as lettuce (Manios et al., 2013) and tomatoes (Asplund and Nurmi, 1991) may contain those bacteria in their tissues (Jr Krzyzanowski et al., 2016).

The risk of the presence of pathogenic bacteria could be aggravated by the presence of antibiotics in wastewater. This increases the number of antibiotic-resistant bacteria. Moreover, the high density of bacteria in WWTP reactors increases the probability of transfer of genetic material between bacteria (Turolla et al., 2018). In Austria, for example, Galler et al. (2018) isolated three multiresistant enterobacteria (extended-spectrum β-lactamase bacteria (ESBL)) from activated sludge: Gram-negative bacilli, methicillin-resistant *Staphylococcus aureus* (MRSA), and Vancomycin-resistant enterococci (VRE)). This could pose sanitary problems because of the dispersion of such antibiotic-resistant bacteria through trophic webs and in the environment (Reinthaler et al., 2013; Fijalkowski et al., 2017; Tozzoli et al., 2017).

The microflora of sewage sludge is also very rich in fungi (Frąc et al., 2014). Fungi play a crucial role in the treatment of wastewater by participating in the degradation of various contaminants (Tozzoli et al., 2017). Several are nevertheless pathogens for plants. For example, two common phytopathogens, *M. circinelloides* and *G. citri-aurantii*, are regularly observed and they affect crop yield by causing diseases in fruits and vegetables. In addition to this ecological and environmental/agronomic risk, with fungi being opportunistic organisms, they have potential pathogenic properties for humans and animals as well (Frąc et al., 2014). Frąc et al. (2017) have found in sewage sludge the fungus *Trichophyton* sp., which is responsible for dermatophytose.

Due to the origin of wastewater, sludge regularly contains viruses, especially of an intestinal origin. Schlindwein et al. (2010) highlighted the most common viruses in WWTP sludge samples from Brazil and tested their viability. The most common viruses were the adenovirus (AdV), the rotavirus (RV), the poliovirus (PV), and finally the hepatitis A virus (HAV). The viability of RV and HAV is around 15%–25% while that of AdV and PV is very high (100% and 90%, respectively), which shows that water and sludge treatment processes are not sufficient to inactivate viruses. This highlights the potential sanitary risks of the dispersal of sludge in the environment.

Furthermore, Bibby and Peccia (2013) found that the most abundant pathogenic viruses were herpes viruses in some US sludge samples. DNA viruses (adenovirus, herpes virus, papillomavirus, and bocavirus) are present in 90% of the samples, and RNA viruses (coronavirus, klassevirus, and rotavirus) are present in 80% of the sludge samples. These viruses can cause serious respiratory and gastrointestinal infections in humans and animals.

Like bacteria, viruses are able to survive once in the environment. Bloem et al. (2017) reported the persistence of enteric viruses for about 100 days in soils.

Works on sewage sludge also reported the presence of parasites such as nematodes and cestodes. Some of them are pathogens for humans and animals and are responsible for various diseases (Chaoua et al., 2017). Sludge frequently contains helminth eggs (*Ascaris*, *Trichuris*, *Toxocara*) (Da Rocha et al., 2016), which are among the most resistant organisms to sludge treatment. Their survival has already been observed for several years after the biosolid to soil application (Bloem et al., 2017).

Other parasites of the protozoan family are also present. Corrêa Medeiros and Antonio Daniel (2018) observed the presence of protozoan cysts in 100% of the samples they controlled and the presence of oocysts in more than half the same samples. A change in sludge treatment had no impact on the concentration and viability of these protozoan forms. Families with some pathogenic species for animals and humans have been observed such as *Cryptosporidium*, Giardia, and Entamoeba (Sabbahi et al., 2018; Khouja et al., 2010).

4 Conclusions and perspectives

Legislative pressure forces all countries to respect the common waste management hierarchy with prevention, reuse, recycling, and recovery the most preferable pathways while landfilling and disposal should be strictly limited (Rorat and Kacprzak, 2017). Authorities, communities, wastewater industries should therefore apply environmental assessments as decision-making tools, in addition to the economic and technical evaluation of each proposed solution. Life Cycle Assessment (LCA) is a tool that allows quantifying the environmental impact/cost of particular options for management of sewage sludge in order to choose the best suitable option for each stakeholder. The result of an LCA shall be understood to be an environmental profile of total and single lifecycle stages considering the use of resources, human health, and ecologic consequences; it does not show any economic or social factors (Cherubini et al., 2009). For instance, the impacts related to wastewater treatment could concern mainly: (1) energy consumption at different stages (global warming), (2) the presence of PTEs (toxicity) and (3) the content of chemical oxygen demand (COD), N and P (eutrophication) (Feijoo et al., 2018). In the case of sewage sludge, most studies examine environmental aspects related to sludge application through fuel requirements (transport on agricultural land), introduction of metals into soils, the reduced use of mineral fertilizers, greenhouse gas emissions, carbon storage, and nutrient leaching (Yoshida et al., 2013). Generally, land application is a contributor to global warming, eutrophication, and acidification while toxicity was considered to be related to the presence of Zn and Cu, according to (Yoshida et al., 2018). Lundin et al. (2004) have compared four different options for final disposal of sewage sludge: agricultural application, coincineration with waste, incineration combined with phosphorus recovery, and fractionation with phosphorus recovery. In most aspects, the agricultural land disposal was recognized as the least preferable from an environmental point of view while other options have good potential of sustainability. Lately, Turunen et al. (2018) have developed a multiattribute value theory (MAVT)-based decision support tool (DST) in order to supply the simple scoring method to count the environmental risks of particular scenarios. The constructed value tree helped to select pyrolysis above the other tested alternatives of composting and incineration.

It is worth noticing that no universal solution can be pointed to as the environmental cost depends on local conditions, which can be highly variable between regions/ countries. Often, the decision-making tools omit the problems of the properties of soil, climate, fauna, and flora present in the environment that can also greatly change the final impact of the sewage sludge on the ecosystems. Unfortunately, the most important decisions considering the treatment of sewage sludge are still being made based on economic and political criteria.

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