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REVIEW PAPER

Composting leachate: characterization, treatment, and future perspectives

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Abstract The increasing production of waste has led to one of the major environmental challenges of today: waste management. A solution to this problem is the composting of organic wastes. While the composting process transforms organic wastes into biologically stable compost, large amounts of highly contaminated leachates that present a direct risk to the environment are also produced. First off, this review discusses the origin and nature of contaminants found in composting leachates. In a general perspective, composting leachates are characterized by the presence of high concentrations of moderately biodegradable organic matter and nutrients and contain toxic pollutants such as heavy metals and plasticizers. Treatment technologies that have been studied are subsequently reported and discussed (treatment efficiencies and operating costs). This review highlights the lack of available solutions to efficiently remove all contaminants found in these leachates, which is a major concern considering the increasing number of composting facilities. While both, membrane bioreactors and reverse osmosis, show promising results with NH_4, COD and TSS removals of > 70, > 85 and $> 99.9\%$, respectively, the resulting effluent remains hazardous for the environment. Further studies are required to assess the use of a combination of biological and advanced oxidation process for the production of a safely disposable effluent.

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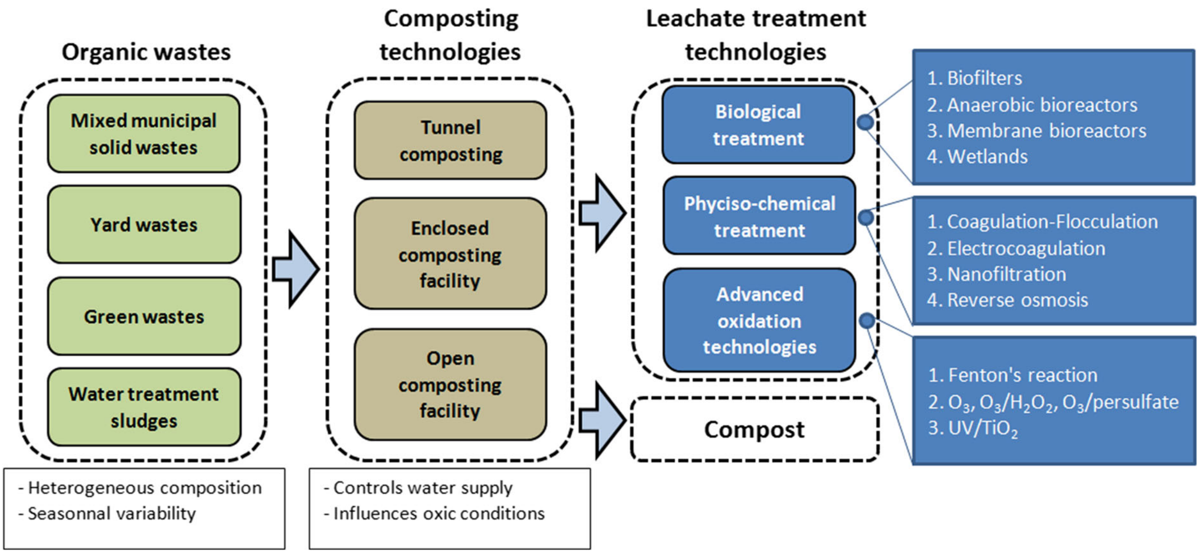
Graphical abstract



Keywords Composting leachate · Composting wastewater · Treatment · Membrane bioreactor · Advanced oxidation

1 Introduction

The exhaustive and exponential consumption of the growing human population has led us to one of the major environmental challenges that we face today: waste management. In 2007, the total amount of waste dumped across the globe was estimated at 2.12 billion tons (UNEP 2009).

Naturally, many different strategies have been proposed in order to attempt to counter and reduce the production of waste, such as composting. Composting technologies are an efficient manner of managing organic waste because they reduce the volume and weight of the initial waste by approximately 50% (Haug 1993; He et al. 2011; Gutiérrez-Miceli et al. 2017). However, composting at the industrial scale generates large volumes leachates. While the compost itself can be reused as a beneficial soil amendment, composting leachates contain various hazardous substances that can have potential adverse effect on the environment, and that therefore need to be sufficiently treated prior to disposal (He et al. 2015; Mokhtarani et al. 2015).

Considering the growing efforts to find solutions to the increase in waste production, the number of composting facilities will most likely increase in the near future. Hence, larger volumes of leachate that needs to be treated will be produced. Research efforts in the field of composting leachate characterization and treatment have subsequently increased as well over the last few decades. Figure 1 illustrates the number of peer reviewed publications in the field of composting leachates since 1991. (Data bases: Google Scholar and Scopus, Keywords: composting leachate, compost leachate, leachate, treatment, organic waste).

Previously, a review has been published on the leachate originating from freshly applied compost in order to regulate the application of fresh compost as soil amendment (Chatterjee et al. 2013). Nonetheless, to the knowledge of the authors, no work has been previously done to summarize the studies conducted on the characterization and treatment of process wastewater originating from the composting process itself (composting leachates). Thus, the main purpose of this review paper is to summarize and discuss with a critical point of view the existing literature on the treatment of composting leachate. To begin, the typical composition of composting leachate is discussed, and the main factors effecting the composition are identified. Following this, the different technologies used for the treatment of composting leachate are outlined, and their efficacy in reducing the

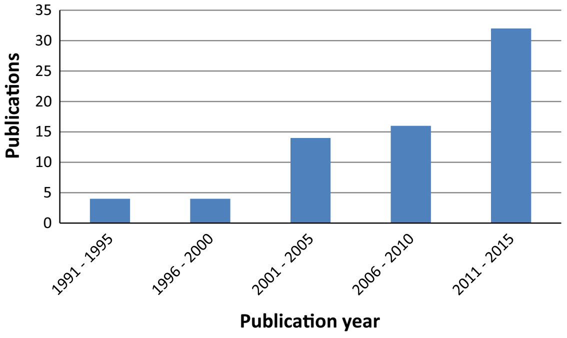


Fig. 1 Graphical representation of peer reviewed publications related to compost leachate since 1991

contamination levels is discussed. Furthermore, these technologies are also compared based on their operating costs. Finally, the future research perspectives in the field of composting leachate characterization, management, and treatment are presented.

2 Composting leachate characterization

2.1 Composting leachate production and contamination

Composting leachates originate from: (1) the water content of the organic waste itself (which represents approximately 60–90% content by weight), (2) the water generated during the composting biochemical reactions, (3) the rain water (open facilities), as well as (4) the water added in order to adjust the moisture content (Krogmann and Woyczechowski 2000). Leachate production is linked to the composting technology, the type of wastes composted, and the climatic conditions. Mixed municipal wastes generate approximately 75–100 L/ton of waste (Liu et al. 2010, 2015; Hashemi et al. 2017), green wastes produce between 5 and 50 L/ton of waste (Cakmakci and Ozyaka 2013; Bakhshoodeh et al. 2017), and sewage composting produces approximately 100 L/ton of sewage (Rajabi and Vafajoo 2012). Generally, composting facilities have the capacity to treat 1000–1500 ton/day of waste with reported leachate productions ranging from 4 to 400 m^3/d (Liu et al. 2010; He et al. 2015; Bakhshoodeh et al. 2017).

Composting leachate contamination mainly occurs during the percolation of water (liquid phase) through the composting of organic waste.

Organic wastes are a major source of biodegradable organic matter, heavy metals, as well as emerging contaminants such as plasticizers (Fig. 2). The presence of a high content of organic matter in leachates results from the breakdown of large and complex organic matter into simpler substances by various microorganisms under aerobic conditions. Initially,

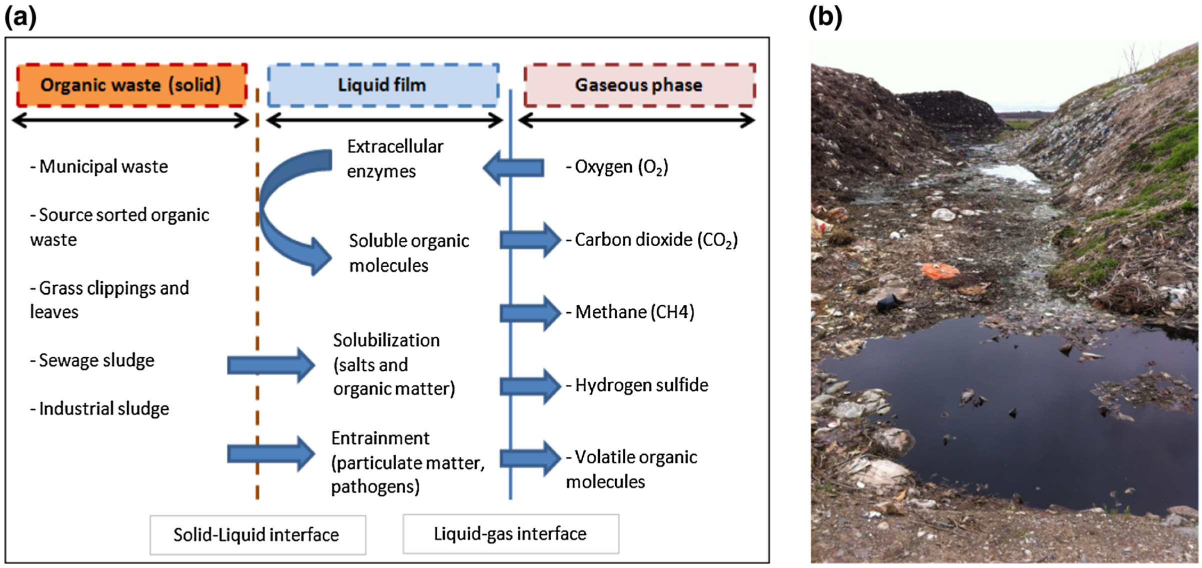


Fig. 2 **a** Schematic representation of the origin of composting leachate contamination, and **b** Composting leachate from an open composting facility

microorganisms already present in the liquid phase surrounding the organic matter utilize the readily available dissolved biodegradable organic matter. Microorganisms simultaneously produce enzymes that react with solid organic matter to release soluble and readily available molecules in the biofilm. In addition to microbial activity, organic and inorganic matter (*i.e.* heavy metals) can be transferred to the liquid phase through solid–liquid mass transfer (de Guardia et al. 2002). Thus, the total mass transfer from biogenic waste to liquid phase can be divided into three different categories: (1) hydrolysis of biogenic waste through enzymatic reactions, (2) solubilization of organic and inorganic molecules and (3) entrainment of particulate matter (Krogmann and Woyczechowski 2000).

2.2 Parameters influencing composting leachate composition

2.2.1 *Feedstock*

The compost source material, *i.e.* the feedstock, determines to a large degree the composition of leachates. Major classes of feedstock suitable for composting have been summarized elsewhere (Chatterjee et al. 2013). Controlling the quality of raw organic waste fed into the composting process can limit the input of materials contaminated with undesired substances, those of which could potentially be found in the resulting composting leachate. Indeed, the presence of heavy metals in composting leachates has a strong correlation to the quality of the composted feedstock (Brown et al. 2013). However, their toxicity, bioavailability, and transport strongly depend on their physico-chemical forms (Bolea et al. 2010). Nutrients (organic and inorganic) concentrations in composting leachate are also influenced by feedstock composition. Several studies were conducted on nutrients availability in compost and results showed changes in chemical composition and biological availability over the course of composting process. Nevertheless, few have studied nutrient concentration in the leachate during the composting process (Amlinger et al. 2003). In addition to influencing the composition of leachates, composted material can also influence the amount of leachate produced. For example, when using wastewater to moisturize solid waste, an excess of substrate can lead to greater generation of leachates (Zahrim et al. 2015).

Seasonal variability of feedstock composition is also an important factor effecting leachate production and composition (Hashemi 2015). Gagnaire et al. (2012) observed leachate production varying from 2 to 14 m^3/day throughout the year due to differences in composted waste (such as water content) in the same composting facility (Gagnaire et al. 2012). García-López et al. (2014) reported, for example, that total nitrogen can vary up to 2 times while copper can vary up to 75 times higher at their peak when compared to their lowest point, due to the seasonal variability of feedstock composition (García-López et al. 2014).

2.2.2 *Composting technology*

The composting technology affects the leachate composition in two different ways: (1) by controlling the water supply in the process, and (2) by influencing the oxic conditions in the composted waste. With tunnel composting technologies, compost is isolated from any external factors, and parameters such as moisture content, odor and aeration are well controlled (Kim et al. 2008). Enclosed and open composting facilities both process organic waste using mechanically-turned windrow (Sanchez-Monedero et al. 2003). In contrast to enclosed and tunnel composting facilities, the composition of composting leachate in an open composting facility is affected by climate conditions (Mullane et al. 2015).

In order to optimize the composting process and accelerate the biodegradation of organic waste, aerobic conditions must be maintained within the wastes. In tunnel composting, aerobic conditions are maintained through both frequent mechanical mixing as well as forced ventilation. However, in open composting facilities, aerobic conditions can only be obtained through mechanical mixing. Thus, open composting facilities operate alternately between aerobic and anaerobic conditions, resulting in a pH drop in leachates due to the rapid conversion of soluble organic molecules to volatile fatty acids (VFAs) in anaerobic conditions (Cho et al. 1995). The influence of the composting technology on composting leachate pH is highlighted in Table 1.

Low pH reported in open composting facility leachates can inhibit the biological nitrification process and make it difficult for biological treatment

Table 1 pH measurements in mixed municipal solid waste composting leachate originating from tunnel composting and open composting facilities

Composting technology	Authors	pH	Average pH
Tunnel composting	Ozkaya (2005)	6.8–7.2	7.4 ± 0.4
	Cakmakci and Ozyaka (2013)	7.1–8.1	
	García-López et al. (2014)	7.4–7.8	
	García-López et al. (2014)	6.8–7.0	
	García-López et al. (2014)	7.4–7.8	
	García-López et al. (2014)	7.3–7.5	
	Romero et al. (2013)	7,0	
	Seyda Özyaka et al. (2015)	7.1–8.1	
Open composting facility	Mokhtarani et al. (2012)	4.2–6.8	5.1 ± 0.3
	Maleki et al. (2009)	4.2–5.5	
	Elyasi, Amani et al. (2015)	4.6–5.2	
	Zazouli and Yousefi (2008)	4.3–5.9	

processes to maintain constant efficacy. Enclosed composting facilities produce a more constant flow and composition of composting leachates due to their controlled environment. However, these facilities use expensive installations and are more costly to operate than open composting facilities.

2.3 Composting leachate composition

Table 2 summarizes the physico-chemical composition and heavy metal concentrations, whereas Table 3 summarizes the nutrient concentrations, both of composting leachate originating from various feedstocks and composting technologies.

In Tables 2 and 3, mixed municipal solid waste (MSW) are composed of all wastes routed to a landfill (Romero et al. 2013), yard wastes are mainly composed of branches, grass clipping, leaves, hedge cuttings, and other similar products, green wastes are composed of both yard wastes and source-separated food wastes (Krogmann and Woyczechowski 2000; Tyrrel et al. 2008) and digestate is the material remaining after the anaerobic digestion of a biodegradable feedstock.

2.3.1 Nutrients: COD, BOD, nitrogen and phosphorus

Table 3 displays the composting leachates' nutrient concentrations (COD, BOD, nitrogen, and phosphorus) reported in the scientific literature. Nitrogen and organic matter are generally considered the two main contaminants found in composting leachates (Romero et al. 2013; Liu et al. 2015). Chemical oxygen demand (COD) is highly variable with reported values varying between 0.08 and 185 g O_2/L. These high concentrations can potentially deplete the oxygen level in the receiving aquatic environment, causing the fatalities of plants, fish, and other aquatic organisms (Hashemi and Khodabakhshi 2016). The average COD concentration is significantly higher in the MSW leachates (48 g O_2/L) than in the yard and green waste leachates (29 g O_2/L). This difference can be explained by the lower biodegradability of the yard and green waste, which is caused by a high proportion of cellulosic biomass.

To assess the biodegradability of this organic contamination in the leachates, the BOD_5/COD ratio is measured. A ratio above 0.5 is considered easily biodegradable and a ratio less than 0.1 is considered difficult to biodegrade (Krogmann and Woyczechowski 2000). The average BOD_5/COD ratio of composting leachate is 0.33, categorizing it as a moderately biodegradable wastewater (Henze et al. 2002). The composting process is characterized by both a disappearance of easily biodegradable compounds as well as an increase in aromatic character. Consequently, low biodegradability of organic matter in composting leachates can be explained by the size and nature of the organic matter in question. Mullane et al. (2015) found that humic- and fulvic-like substances with high molecular weight formed during composting are responsible for an important part of organic contamination in composting leachates (Mullane et al. 2015). He et al. (2015) reached the same conclusion by findings significant fluorescence peaks

Table 2 Composting leachate characterization: physico-chemical parameters and heavy metals

Parameters	pH	Conductivity (mS/cm)	TSS g/L	TVS g/L	Cl mg/L	Cd mg/L	Cu mg/L	Ni mg/L	Pb mg/L	Zn mg/L	Authors
WHO Guidelines (aquatic life)	6.5–9^{a}		see^{b}	–	230^{a}	0.00072^{a}	0.0065 – 0.034^{c}	0.052^{a}	0.0025^{a}	0.12^{a}	
1. Mixed municipal solid wastes											
Tunnel composting	6.8–7.2	–	0.35–0.4	–	–	–	–	–	–	–	Ozkaya (2005)
	7,0	–	–	–	–	0.01	<0.01	0.05	0.02	0.5	Romero et al. (2013)
	7.1–8.1	21–25	1.77–1.87	–	2,910–3,210	–	–	–	–	–	Cakmakci and Ozyaka (2013)
	7.4–7.8	8.4–8.8	1.86–2.26	1.78–2.22	–	0.13	0.2	0.44	0.85	2.3	García-López et al. (2014)
	6.8–7.0	35.3–36.7	8.38–11.9	13.9–14.4	–	0.17	1.5	1.47	1.07	7.8	García-López et al. (2014)
	7.4–7.8	1.9–2.9	18.0–18.3	6.03–6.13	–	0.03	1	0.61	0.22	3.3	García-López et al. (2014)
	7.3–7.5	1.4–2.2	0.35–23.4	9.2–10.9	–	0.04	0.6	0.79	0.22	8.2	García-López et al. (2014)
	7.1–8.1	13–17	1.75–1.90	–	2,910–3,210	–	–	–	–	–	Seyda Özyaka et al. (2015)
Enclosed composting facility	5.9–7.9	23.3–48.1	4.7–25.8	2.6–11	–	0.05	0.8	0.75	1.02	1.32	Rajabi and Vafajoo (2012)
	4.4	33.5	–	–	–	–	–	–	–	–	Hashemi et al. (2017)
Open composting facility	7.0–7.4	3.2–5.8	0.22–0.74	–	–	–	–	–	–	–	Laitinen et al. (2006)
	6.2	51.1–52.1	–	–	–	–	–	–	–	–	Trujillo et al. (2006)
	6.6–6.9	–	0.05	–	–	–	–	–	–	–	Zhang et al. (2007)
	4.3–5.9	–	10.8–24	15.7–26.2	–	0.06–1.31	0.34–2.34	0.72–4.01	–	1.26–12.7	Zazouli and Yousefi (2008)
	4.2–5.5	–	–	–	–	0.07–0.34	0.32–2.25	0.65–3.8	–	1.2–11	Maleki et al. (2009)
	4.2–6.8	20–56	2.9–17	2.3–65	–	–	–	–	–	–	Mokhtarani et al. (2012)
	4.6–5.2	–	1.92	–	–	–	–	–	–	–	Elyasi, Amani et al. (2015)
	4.7–5.1	27.5–30.3	–	–	–	–	–	–	–	–	Bakhshoodeh et al. (2017)
-	3.8–6.3	–	16–33.7	–	3,589–8,667	0.14–1.58	0.11–0.49	0–1.19	0.06–0.95	1.7–37.5	Liu et al. (2010)
	4.9	–	9.03	–	295	1.24	–	4.43	4.28	–	Tabatabaei et al. (2012)
	–	–	–	–	–	–	0.32	0.43	–	15.8	Liu et al. (2015)

Table 2 continued

Parameters WHO Guidelines (aquatic life)	pH $6.5–9^{a}$	Conductivity (mS/cm)	TSS g/L see^{b}	TVS g/L –	Cl mg/L 230^{a}	Cd mg/L 0.00072^{a}	Cu mg/L 0.0065 – 0.034^{c}	Ni mg/L 0.052^{a}	Pb mg/L 0.0025^{a}	Zn mg/L 0.12^{a}	Authors
2. Yard wastes											
Enclosed composting facility	7.2–8.2	9.4–27.9	–	–	1,514–5,254	–	–	–	–	–	Krogmann and Woyczechowski (2000)
Open composting facility	8.4–9.2	3.9–6.4	–	–	–	<0.1	0.25–0.38	–	<0.25	–	Cole (1994)
	6.2–8.5	–	–	–	–	–	–	–	–	–	Krogmann and Woyczechowski (2000)
Lab-scale composting	5.0–7.4	10	–	–	–	–	–	–	–	–	Krogmann and Woyczechowski (2000)
3. Green wastes											
Enclosed composting facility	7.6	5.9	–	–	780	0.01	0.31	0.21	0.03	0.79	Ulén (1997)
	5.1	82.6	–	–	3,720	0.02	0.11	0.59	0.81	13.5	Brown et al. (2013)
Open composting facility	6.9–8.6	–	0.08–0.6	0.1–0.43	–	–	–	–	–	0.2	Lafrance et al. (1996)
	7.9	6	–	–	1,150	–	0.19	0.13	–	0.3	Ulén (1997)
	4.6–5.8	–	2.09–2.46	–	–	–	–	–	–	–	Savage and Tyrrel (2005)
	8.6–8.8	2.9–4.1	0.09–0.19	–	–	–	–	–	–	–	Tyrrel et al. (2008)
	4.7–5.6	9.2–16.9		–	975–1,168	0.3–0.98	<0.01–0.24	0.33–0.38	0.004–0.06	<0.01–3.37	Justin et al. (2010)
	3.4	–	–	–	–	0.005	0.036	0.041	0.033	0.34	Zhou et al. (2010)
Lab-scale composting	3.0–9.2	8.6	–	–	–	–	–	–	–	–	Krogmann and Woyczechowski (2000)
–	8.8	–	< 0.001	–	–	–	–	–	–	–	Liu and Lo (2001)
4. Wastewater treatment sludge											
Enclosed composting facility	–	2.7–78	0.11–14.4	0.09–10.3	–	–	–	–	0–0.52	0.21–15	Gagnaire et al. (2012)
Open composting facility	7.1	46.7–47.5	–	–	–	–	–	–	–	–	Trujillo et al. (2006)
–	6.3–8.0	9.5–78	0.11–1.7	0.09–1.46	32–1,030	<0.002	<0.1–0.38	0.12	<0.10–0.17	0.29–14	Gagnaire et al. (2011)
5. Digestate											

Table 2 continued

Parameters	pH	Conductivity (mS/cm)	TSS g/L	TVS g/L	Cl mg/L	Cd mg/L	Cu mg/L	Ni mg/L	Pb mg/L	Zn mg/L	Authors
WHO Guidelines (aquatic life)	6.5–9[a]		see[b]	–	230[a]	0.00072[a]	0.0065 – 0.034[c]	0.052[a]	0.0025[a]	0.12[a]	
–	7.9–8.1	9.3- 23.2	2.59–6.21	–	–	–	–	–	–	–	Zuriaga-Agustí et al. (2016)
6. Undefined											
–	7.4–8.0	5.3–23.4	0.01–11.5	5–13	–	–	–	–	–	–	Amani et al. (2014)
	6.2–8	2.1–20.7	–	–	–	–	–	–	–	–	Hashemi et al. (2015)
	–	–	0.13–2.7	–	–	–	–	–	–	–	Hashemi 2015)

[a] National Recommended Water Quality Criteria—Aquatic Life Criteria Table (Agency 2017)

[b] Quality Criteria for Water—1986 (Agency 1986)

[c] Water Quality Standards Handbook: Second edition (Agency 1994)

in the 300–380 and 380–500 nm regions, which corresponds to fulvic- and humic like substances, respectively (He et al. 2015). Generally, the molecular sizes of aquatic fulvic- and humic-acids are in the ranges 500–2000 and 2000–5000 Da, respectively (Nissinen et al. 2001). These organic molecules are known to play a significant role in both the organic matter's resistance to biodegradation, as well as heavy metal transport within the environment through complexation (He et al. 2011; Mullane et al. 2015).

Nitrogen leached during the microbial decay of organic matter is mainly found as dissolved organic nitrogen comprised of: (1) proteinaceous compounds and amines integrated into the fulvic- and humic-like substances and (2) ammonium salts(Popa and Green 2012; He et al. 2015). From Table 3, total Kjeldahl Nitrogen (TKN) concentrations measured in composting leachates range from 7.2 to 18 570 mg N/L.The proportion of organic nitrogen found as ammonium (NH_4/NTK ratio) is 0.57 ± 0.19 mg N-NH_4/mg N_{org}, which is comparable to domestic sewage in terms of bioavailability (Tas et al. 2009). US-EPA recommended TKN criteria for rivers and streams vary between 0.12 and 2.18 mg N/L, which is more than 1000 times lower than the concentrations found in composting leachates.

From Table 3, total phosphorus concentrations measured from different leachates range from 0.52 to 485 mg P/L. This phosphorus mainly originates from the decomposition of easily biodegradable organic wastes such as MSW and green wastes. The highest phosphorus concentration was measured in a leachate originating from green wastes, composted in an enclosed facility (485 mg P/L) (Brown et al. 2013). While phosphorus does not pose any direct risk to aquatic life, an increased phosphorus concentration in aquatic environments incites the proliferation of algae and cyanobacteria, which in turn accelerates the eutrophication of aquatic environments. US-EPA recommended TP criteria for rivers and streams vary between 0.01 and 0.128 mg P/L, which is also more than 1000 times lower than average concentrations of composting leachates.

2.3.2 Heavy metals

Table 2 presents the composting leachates' heavy metal concentrations reported in the scientific literature. Average concentrations of the measured heavy metals are: 0.36 mg Cd/L, 0.61 mg Cu/L, 1.06 mg Ni/

Table 3 Composting leachate characterization: nutrients and organic matter

Parameters	COD g O^2/L	BOD_5 g O^2/L	BOD_5/COD ratio	TKN mg N/L	NH_4 mg N-NH_4/L	TP mg/L	PO_4 mg P-PO_4/L	Authors
WHO Guideline (aquatic life)	–	–	–	0.12 – 2.18[b]	1.9[a]	0.01 – 0.128[b]	–	
1. Mixed municipal solid wastes								
Tunnel composting	8.5–9.0	–	–	350–400	250–300	–	30–50	Ozkaya (2005)
	14.7–15.7	–	–	1100	645–655	–	–	Romero et al. (2013)
	22.9–24.9	–	–	1445–1545	705–805	–	–	Cakmakci and Ozyaka (2013)
	10.6–12.6	1.22–1.29	0.11	1445	730–800	–	81–85	García-López et al. (2014)
	38.8–42.8	0.67–0.72	0.02	2853	2,250–2275	–	80–84	García-López et al. (2014)
	24.6–25.6	7.59–7.90	0.33	1612	950–1015	–	87	García-López et al. (2014)
	36.8–38.3	5.70–5.93	0.17	3213	1550–1625	–	12	García-López et al. (2014)
	–	–	–	1445–1545	–	–	–	Seyda Özyaka et al. (2015)
Enclosed composting facility	42.2– 107.2	40.1	–	758	520	0.52	–	Rajabi and Vafajoo (2012)
	95.5	55.2	0.58	2,300	–	280	–	Hashemi et al. (2017)
Open composting facility	1.97–2.43	1.0–1.6	–	160–320	120–300	8.2–13.4	1.4–5.6	Laitinen et al. (2006)
	169–185	85–89	0.49	–	–	–	–	Trujillo et al. (2006)
	25.8–45.5	–	–	–	–	–	–	Zazouli and Yousefi (2008)
	0.44–0.48	–	–	43.6–49.2	36–43	–	–	Zhang et al. (2007)
	22.3–45	–	–	–	–	–	–	Maleki et al. (2009)
	51–109	23–55	0.42–0.53	467–1860	258–1290	32–193	–	Mokhtarani et al. (2012)
	23–40	–	–	–	–	–	–	Elyasi, Amani et al. (2015)
	102.8–107.9	67.0–71.6	–	5,820–6,660	275–360	–	–	Bakhshoodeh et al. 2017)
–	51.2–71.3	26.2–30.4	–	1,418–2,967	231–796	104–288	–	Liu et al. (2010)
	100	–	–	–	–	–	–	Tabatabaei et al. (2012)
	55.7	–	–	10,056	–	71	–	Liu et al. (2015)
2. Yard wastes								
Enclosed composting facility	2.43–31.8	0.008–11.6	–	–	98–558	–	–	Krogmann and Woyczechowski (2000)
Open composting facility	–	–	–	–	5.1–10.5	–	–	Cole (1994)
Lab-scale composting	0.08	0.03	0.22	7.2	1.3	–	–	Krogmann and Woyczechowski (2000)

Table 3 continued

Parameters	COD g O^2/L	BOD_5 g O^2/L	BOD_5/COD ratio	TKN mg N/L	NH_4 mg N-NH_4/L	TP mg/L	PO_4 mg P-PO_4/L	Authors
WHO Guideline (aquatic life)	–	–	–	0.12 – 2.18[b]	1.9[a]	0.01 – 0.128[b]	–	
3. Green wastes								
Enclosed composting facility	–	–	–	240	–	28	–	Ulén (1997)
	116	–	–	–	2720	485	–	Brown et al. (2013)
Open composting facility	0.85–10	0.15–5.4	0.18–0.87	40–150	4–71	28–35	–	Lafrance et al. (1996)
	–	–	–	420	–	16	–	Ulén (1997)
	–	44.7–52.8	–	–	–	–	–	Savage and Tyrrel (2005)
	0.53–1.15	0.02–0.26	0.03–0.25	–	6.5–24.1	1.5–2.4	–	Tyrrel et al. (2008)
	–	–	–	970–1105	887–980	46–372	15–119	Justin et al. (2010)
	75.7	–	–	–	824	–	–	Zhou et al. (2010)
Lab-scale composting	0.29	0.06	0.26	250	107	–	–	Krogmann and Woyczechowski (2000)
–	18–68	10–46	0.55–0.67	–	400–1,100	–	–	Krogmann and Woyczechowski (2000)
	0.98	0.1	0.11	–	–	–	–	Liu and Lo (2001)
4. Wastewater treatment sludge								
Enclosed composting facility	1.46–43.4	0.36–17.4	0.2–0.59	345–18,570	218–21,180	7–88	–	Gagnaire et al. (2012)
Open composting facility	126–140	70–72	0.54	–	–	–	–	Trujillo et al. (2006)
–	1.88–42.4	1.6–17.4	0.32–0.55	1470–7960	1167–21,180	22–78	–	Gagnaire et al. (2011)
5. Digestate								
–	1.88–4.40	–	–	470–1110	443–1091	–	–	Zuriaga-Agustí et al. (2016)
6. Undefined								
–	83–107	–	0.2–0.3	–	–	–	–	Amani et al. (2014)
	0.42–7.25	0.07–2.3	0.28–0.38	–	–	–	–	Hashemi et al. (2015)
	0.085–5.36	0.1–0.5	–	–	–	–	–	Hashemi (2015)

[a] National Recommended Water Quality Criteria—Aquatic Life Criteria Table (Agency 2017)

[b] Summary Table for the Nutrient Criteria Documents (Agency 2014)

L, 0.64 mg Pb/L and 6.93 mg Zn/L. The highest concentrations for each of these metals were found in the leachates originating from MSW composted in open composting facilities (Table 2) due to the low pH caused by anoxic conditions within the organic wastes which increases the mobility of the heavy metals contained in the miscellaneous wastes from the MSW.

When compared to the NRWQC criteria for freshwater aquatic life, each metal has a concentration of at least ten times higher that of the recommended concentration. To assess the bioavailability and mobility in natural environments, metal complexation with organic matter in composting leachates has been studied. He et al. (2015) observed that Ni exhibited a strong complexation capability with aliphatic structures, carbohydrates and proteinaceous compounds, although there was no significant complexation between proteinaceous compounds and amine with Cu, Cd and Zn (He et al. 2015). Weng et al. (2002) showed that Cu and Pb have higher tendencies to form complexes with natural organic matter than do Cd, Zn and Ni (Weng et al. 2002). Metal binding to organic matter in composting leachates can affect the solubility of metals to a considerable extent. The formation of organometallic complexes, combined with low pH leachates and the presence of large humic- and fulvic-like molecules, can explain the high concentrations of heavy metals found in composting leachates.

2.3.3 Dissolved and suspended solids

Dissolved and suspended solid concentrations in composting leachates are presented in Table 2. The conductivity measurements in composting leachates range from 1420 to 82 600 µS/cm. For comparison, the typical conductivity of potable freshwater and seawater are estimated at 30–1500 µS/cm and 53,000 µS/cm, respectively (Assiry et al. 2010, Fondriest Environmental 2014). Instead of regulating the conductivity, the NRWQC sets a chloride concentration criterion at 230 mg/L for freshwater aquatic life. Chloride concentrations measured in composting leachates range from 32 to 8667 mg/L (Table 2). Thus, chloride in composting leachate can reach concentrations ten times higher than the criterion. Concentration of total suspended solids (TSS) in composting leachates is also highly variable (86–33 680 mg/L). This variability can be caused by the leachate collection system (residence time, turbulence) and the particle size. However, this particular information has not been discussed in any of the studies. In comparison to municipal wastewater TSS concentrations, which generally varies between 100 and 350 mg/L with an average of 220 mg/L, composting leachate TSS concentration is very high and is likely to cause a decrease of light availability which can reduce photosynthetic rates, leading to adverse effects on algae and macrophytes (Quinteiro et al. 2015).

2.3.4 Toxic organic pollutants

With the increase in number of composting facilities, as well as the variety of composted wastes, the presence of toxic organic pollutants (TOPs) in composting leachates is worth paying attention to (Kuster et al. 2008; Lonappan et al. 2016). Research conducted by Marttinen et al. (2004) and Fromme et al. (2002) have evidenced the presence of phtalates (a common plasticizer) in composting leachates (Fromme et al. 2002; Marttinen et al. 2004). Fromme et al. (2002) reported concentrations of 17–26 µg DEHP/L and 25–146 µg BPA/L in mixed municipal-solid wastes composting leachates. These plasiticizers may originate from the plastic bags in which organic wastes are collected. Marttinen et al. (2004) reported concentrations of 9–34 µg DEHP/L in municipal sewage sludge composting leachates (Marttinen et al. 2004). Bis-phenol A (BPA) and polycyclic aromatic hydrocarbons (PAHs), such as pyrene, have also been detected in composting leachates (Zhang et al. 2007; Hashemi et al. 2016a, b).

Due to the significant number of sources from which TOPs can be introduced into the composting process, the possible leachate TOPs contamination should be considered when assessing leachate treatments.

3 Composting leachate management and treatment

3.1 Reducing leachate generation

The least expensive option for leachate management is to design the composting facility in such a way that reduces and reuses all excess water, meaning that leachate treatment is only needed as a last resort. In order to prevent leachate production, the facility should be designed to separate rainfall and snowmelt from organic wastes and compost. Then, a proper mix of compost feedstocks should be used; this ideal mix of compost feedstocks includes neither using materials that are too wet nor adding to much water to the mixture. To reach this objective, the moisture content of the compost should never exceed 65% (Forgie et al. 2004).

Similarly to bioreactor landfill, leachates generated during the composting process can be reintroduced in the compost windrow to provide humidity control and process optimization. Bilgili et al. (2007) studied the effect of recirculating leachates in lysimeter filled with MSW. Recirculating leachates reduced the total amount of leachate produced by 47.1%, and reduced the initial concentration of COD (40 g O_2/L) by 93% (Bilgili et al. 2007). Thus, recirculating the composting leachates both reduces the volume and the charge of organic contaminants to be treated, as well as improves the composting process (Ming et al. 2008).

Excess leachates that cannot be reused must be treated up to a standard that does not impact the environment. Different treatment technologies have been developed or specifically adapted for the treatment of composting leachates. In order to discuss their efficacy in reducing composting leachate contamination, these technologies have been separated into three different categories: (1) Biological treatment, (2) Physico-chemical treatment, and (3) Advanced technologies. Furthermore, the potential reuse or transformation of composting leachate, as well as the strategies to reduce the amount of leachates produced, are also discussed.

3.2 Biological treatment

The lower operating costs and simple operation of biological wastewater treatment technologies, compared to the physico-chemical and advanced treatment technologies, makes biological treatment technologies attractive for composting facilities that need low-cost options for leachate treatment. However, despite these advantages, biological treatments are known to require extensive hydraulic retention time (HRT $>$ 48 h), and to produce large amounts of sludge that need to be disposed off. The selection of appropriate biological treatment is based on the C_{BOD5}/N/P ratio. For a of 100/5/1 ratio, an aerobic biological treatment is generally recommended (Lafrance et al. 1996). For 250 to 500/5/1 ratio, an anaerobic treatment/digestion is generally recommended (Mokhtarani et al. 2012). Calculated from Table 3, the average C_{DBO5}/N/P ratio in composting leachate is 320/30/1. Thus, the C/P ratio of composting leachates fulfills the requirements for anaerobic treatment while the C/N ratio fulfill those for an aerobic treatment. The treatment efficacies of the different biotreatment technologies applied to composting leachates are summarized in Table 4.

3.2.1 *Biofilters*

Biofilters are packed-bed reactors filled with different media in which composting leachate is percolated. The media both acts as a support for microbial growth and increases the contact surface area between the biofilm and the leachates. A preliminary study was conducted by Lafrance et al. (1996) using mature compost, peat and vermiculite as media. The best results were obtained with a 35/35/30 mixture of peat, mature compost and vermiculite (Lafrance et al. 1996). Using this mixture, they removed 95% of BOD_5, 48% of COD, and 57% of ammoniacal nitrogen. Subsequently, Savage and Tyrrel (2005) conducted a laboratory study using six different packing materials: polystyrene, soil, broken bricks, mature compost, oversize (woody waste produced in the final screening of compost), and wood mulch (Savage and Tyrrel 2005). The best results were obtained with the oversize media with a BOD_5 and ammoniacal nitrogen average removal of 78%. Both studies concluded that biofilters made a significant impact on composting leachate quality, however, the filtrate remained highly polluted and required further treatment (Lafrance et al. 1996; Savage and Tyrrel 2005).

Following these lab-scale studies, Tyrrel et al. (2008) conducted a pilot scale study using two types of filter media, mature compost and oversize. Applying hydraulic loading rates ranging from 0.05 to 0.5 m^3/m^3(media)/day, they produced an effluent with $<$ 10 mg/L ammoniacal nitrogen on $>$ 95% of sampling occasions (initial concentrations of ammoniacal nitrogen ranged from 6.5 to 32 mg/L) with the oversize media. However, they obtained low COD removal results (Tyrrel et al. 2008).

In general, biofilters with oversize material as a media are an efficient treatment technology for NH_3 removal. The advantage of this technology is that is reuses available residues that have no/low-cost for composting facilities. However, hydraulic loading rates are low, meaning that the size of such a system would be exceedingly large for large scale composting facilities. Furthermore, COD removals are deficient, meaning that further treatments would be needed. A

Table 4 Summary of contaminants removal efficiency using biological treatment for composting leachate treatment

Configuration	TSS		COD		BOD_5		TKN		NH_4		Authors
	Feed (mg/L)	Removal (%)	Feed (g/L)	Removal (%)	Feed (g/L)	Removal (%)	Feed (mg/L)	Removal (%)	Feed (mg/L)	Removal (%)	
1. Biofilters											
Column (peat/compost/ vermiculite)	80–600	36–95	0.85–10	33–70	0.15–4.5	62–96	40–150	60	4–71	35–57	Lafrance et al. (1996)
Column (sieving oversize)	86–192	–	0.53–1.15	9 -24	0.02–0.26	68–92	–	–	6.5–32.1	75–86	Tyrrel et al. (2008)
Column (sieving oversize)	2,275 ± 184	–	–	–	48.7 ± 4	34–78	–	–	705 ± 23	31–78	Savage and Tyrrel (2005)
2. Anaerobic bioreactors											
Expanded granular sludge bed bioreactor (EGSB)	15,961–33,682	70–85	51.2–71.3	88–97	–	–	1,418–2,967	–	231–796	0	Liu et al. (2010)
Fixed bed down flow–up flow anaerobic bioreactor	–	–	81 ± 14.5	85	39 ± 10	89	–	–	–	–	Mokhtarani et al. (2012)
Anaerobic continuous stirred tank reactor (ACSTR)	–	–	15.2 ± 0.5	87	–	–	1,100	23	650 ± 6	0	Romero et al. (2013)
Sequencing batch reactor (ASBR)	–	–	83–107	71–74	–	–	–	–	–	–	Amani et al. (2014)
Baffled reactor	–	77	8.3	84	–	–	–	–	–	–	Amani et al. (2015)
Sequencing batch reactor (ASBR)	–	–	11.2 ± 8.6	71–81	–	–	–	–	–	–	Hashemi et al. (2015)
Sequencing batch reactor (ASBR)			95.5	79.5–84	55.2	87–91	2,300	74			Hashemi et al. (2017)
Anaerobic migrating blanket reactor (AMBR)			8.5	97.4	3.25	99.4					Eslami et al. (2018)
3. Membrane bioreactors											
Sequencing batch reactor/submerged membrane	480 ± 260	99.8	2.2 ± 0.2	85	1.3 ± 0.3	97	240 ± 80	–	210 ± 90	97	Laitinen et al. (2006)
Aerobic reactor/submerged membrane	–	–	116	99.7	–	–	–	–	2,720	99.9	Brown et al. (2013)
Aerobic reactor/submerged membrane	–	–	10.6–42.8	20–90	–	–	1,445–3,213	40	730–2,275	21	García-López et al. (2014)

Table 4 continued

Configuration	TSS		COD		BOD_5		TKN		NH_4		Authors
	Feed (mg/L)	Removal (%)	Feed (g/L)	Removal (%)	Feed (g/L)	Removal (%)	Feed (mg/L)	Removal (%)	Feed (mg/L)	Removal (%)	
Sequencing batch reactor/submerged membrane	125–2,700	99.9	0.09–5.4	70	0.1–0.5	93	–	–	–	–	Hashemi (2015)
Sequencing batch reactor/submerged membrane	120–2,800	99.9	0.14–4.2	70	0.1–0.63	93	–	–	–	–	Hashemi and Khodabakhshi (2016)

summary of the treatment efficacies of biofilters is presented in Table 4.

3.2.2 Anaerobic bioreactors

Anaerobic bioreactors (AB) are an assembly of processes involving biological decomposition of organic matter under anaerobic conditions. Anaerobic digestion has attracted increasing attention, mainly due to its considerable benefits such as usable biogas, high refractory COD treatment efficiency and low sludge production (Elyasi, Amani et al. 2015). Studies on composting leachate treatment have reported the use of both simple anaerobic systems (anaerobic continuous stirred tank reactor (A-CSTR) (Romero et al. 2013), anaerobic baffled reactor (Elyasi, Amani et al. 2015) and anaerobic sequencing batch reactor (ASBR) (Amani et al. 2014; Hashemi et al. 2015, 2017)), as well as more complex anaerobic systems (expanded granular sludge bed bioreactor (EGSB) (Liu et al. 2010), anaerobic migrating blanket reactor (AMBR) (Hashemi et al. 2016a, b; Eslami et al. 2018) and fixed bed down flow-up flow anaerobic bioreactor (Mokhtarani et al. 2012). Treatment efficiencies of these bioprocesses are summarized in Table 4.

COD removals using AB vary between 71 and 99.4%, with the anaerobic migrating blanket reactor having the highest removal rates reported (82.4–97.5% (Liu et al. 2010; Eslami et al. 2018). Mokhtarani et al. (2012) studied the effect of different OLRs on COD removal efficiency, and found that COD removal efficiency was independent of OLR below a value of 6 kg COD/m^3/day. Above that OLR, the COD removal efficiency was significantly reduced (Mokhtarani et al. 2012). However, these findings disagree with those of Liu et al. (2010), who found that an OLR of 22.5 ± 1.5 kg COD/m^3/day was optimal. While both organic contaminations have similar BOD_5/COD ratios (ranging from 0.42 to 0.53), potential explanations for the discrepancy between their conclusions include different Food/Microorganisms (F/M) ratios which were not discussed.

Mokhtarani et al. (2012) observed that although nutrient concentrations in their raw leachate (C/N/P = 350/4/0.4) were low, they had no significant effect on the COD removal efficiency and performance of the reactors (Mokhtarani et al. 2012). This particularity of biological treatment is interesting for

Table 5 Biogas production potential of composting leachates

Anaerobic bioreactor technology	BOD_5/COD ratio	Yield (L biogas/g COD)	Biogas quality (% methane)	Conversion (% COD)	Authors
Expanded granular sludge bed bioreactor (EGSB)	0.43–0.51	0.23–0.29	60–80	80–83	Liu et al. (2010)
Anaerobic sequencing batch reactor (ASBR)	0.2–0.3	0.5–0.52	75	–	Amani et al. (2014)
Anaerobic sequencing batch reactor (ASBR)	0.28–0.38	0.36	55–65	85	Hashemi et al. (2015)
Anaerobic migrating blanket reactor (AMBR)	0.38	0.34	–	–	Eslami et al. (2018)

wastewaters with variable composition, such as composting leachates.

Hashemi et al. (2016a, b) studied the removal of PAHs and heavy metals using an anaerobic migrating blanket reactor. With OLR ranging from 1 to 19.7 g COD/L/day, they obtained an average removal efficiency of 73% for the PAHs. In terms of the heavy metals (Ni, Zn, Cr, Cd, Pb, Cu), the AMBR process reduced their concentrations by 42–55% (Hashemi et al. 2016a, b).

Conversion potential of the organic contamination to biogas has also been studied (Liu et al. 2010, Amani et al. 2014, Hashemi et al. 2015, Eslami et al. 2018). The yield and the quality of biogas produced are presented in Table 5.

Conversion rates are considered high, with reported values of 80–85%, despite the low biodegradability of organic contamination ($BOD_5/COD = 0.28–0.38$). The methane content of biogas originating from the anaerobic digestion of composting leachate is comparable to the anaerobic digestion of household wastes (50–60%) and agricultural wastes (60–75%). Using an AMBR, Eslami et al. (2018) showed that the optimal production of biogas is obtained with an OLR of 10 g $COD \times L^{-1} \times day^{-1}$. Higher OLRs lead to a sudden decrease in biogas production due to the accumulation of VFAs that become harmful for methanogenic Achaea bacteria (Eslami et al. 2018). Whereas studies have proven that AB is efficient for COD removal, this process is not as efficient for ammoniacal nitrogen contamination. Liu et al. (2010) noted that ammoniacal nitrogen concentration in the effluent was 150–400 mg/L higher than in the influent. Their explanation was that proteins in the influent are biodegraded and some nitrogen is converted in the form of NH_3 (Liu et al. 2010). This finding is further supported by Romero et al. (2013), who also observed an increase in ammoniacal nitrogen concentration after the anaerobic digestion of raw leachates (Romero et al. 2013). Thus, the presence of refractory COD, TSS, high turbidity, and high concentrations of ammoniacal nitrogen in anaerobic reactor effluents confirms the necessity of other techniques as post-treatments.

3.2.3 *Membrane bioreactors*

An alternative technology to anaerobic bioreactors, known for its high COD removal efficiency, its small foot print and its capacity of treating wastewater with lower biodegradability, is the membrane bioreactor (MBR). MBRs are wastewater treatment processes that integrate a permselective membrane with a biological process (Judd 2010). The membrane serves as an advanced treatment unit for complete TSS removal and as a barrier to microorganisms (Hashemi 2015, Zuriaga-Agustí et al. 2016). However, MBRs are still facing a major operating problem: membrane fouling (Hashemi et al. 2016a, b; Zuriaga-Agustí et al. 2016). Hashemi et Khodabakhshi reported that the use of granular activated carbon within a MBR reduces membrane fouling while also removing a significant level of recalcitrant and the biorefractory compound found in composting leachates (Hashemi and Khodabakhshi 2016). Studies on composting leachate treatment using MBRs have reported the use of different combinations of bioreactors and membrane configurations: sequential bioreactors (SBR) (Laitinen

Table 6 Operating conditions of MBRs applied to composting leachate treatment

Membrane bioreactor configuration (Biological process/membrane module)	MLSS (g/L)	MLVSS (g/L)	HRT (h)	SRT (days)	OLR (kg COD/m^3/day)	Authors
SBR/submerged	–	–	72	35–60	–	Laitinen et al. (2006)
SBR/submerged	4–11	–	16	–	0.13–8.03	Hashemi (2015)
SBR/Submerged	4–11		12–23	–	0.01–10.7	Hashemi and Khodabakhshi (2016)
Aerobic-anaerobic/external	13.1–27.8	10.1–18.7	–	–	–	Zuriaga-Agustí et al. (2016)
Aerobic/submerged	–	–	2.3	–	1.2	Brown et al. (2013)

et al. 2006; Hashemi 2015; Hashemi and Khodabakhshi 2016) and aerobic tank (Brown et al. 2013; García-López et al. 2014) (both combined with submerged membranes), and aerobic-anaerobic bioreactors combined with external membranes (Zuriaga-Agustí et al. 2016). Composting leachate treatment efficiencies for each of these MBR configurations are summarized in Table 4. Operating conditions for MBRs applied to composting leachate treatment are presented in Table 6.

As expected with membrane filtration processes, MBRs TSS removals range from 99.8 to 99.9%. The aerobic tank/submerged membrane configuration have the highest COD and ammoniacal nitrogen removal performances (99.7% and 99.9%, respectively) (Brown et al. 2013). While these removal efficiencies are promising for the use of MBRs for composting leachate treatment, Brown et al. (2013) operated their MBR under impracticable conditions: the hydraulic retention time (HRT) was unrealistically high at 2289 h (95 days). Furthermore, the MBR was only fed during 39 days, which means that the authors have never completed a single hydraulic residence time (95 days) during their experiment.

In addition to having few studies conducted on composting leachate treatment using MBRs, little information is available on the operating conditions. Operating parameters, such as mixed liquor suspended solids concentration (MLSS), HRT, and solid retention time (SRT) have direct effects on MBR performances, and need to be reported when assessing the performances of MBR. Thus, there is a significant lack of disclosing of information regarding the use of MBR for composting leachate treatment.

3.2.4 *Wetlands*

Due to the fertilizing properties of composting leachates, Bakhshoodeh et al. (2017) studied their treatment by subsurface horizontal flow constructed wetland. Five months after planting Vetiveria zizanioides, results showed significant removal efficiencies (74.5% of BOD5, 53.7% of COD, and 69.9% of N-NH3). While this management method is limited to warmer climates, it has the potential to be used as a leachate pretreatment (Bakhshoodeh et al. 2017).

3.3 Physico-chemical treatment

Several studies reported the use of different physico-chemical treatment technologies. These studies are reported here in two categories: (1) Coagulation-flocculation, and (2) Membrane filtration. The summary of their treatment efficiencies are reported in Table 7.

3.3.1 *Coagulation-flocculation*

Coagulation and flocculation processes are efficient for the removal of particulate fractions of wastewaters (particular matter, colloids, clays, etc.), high molecular weight organic fractions (humic acids, fulvic acids) and phosphorus.

Mahvi et al. (2015) tested different coagulants (ferrous sulfate, ferric chloride, poly ferric sulfate, alum and poly aluminum chloride), along with two anionic and cationic commercial co-coagulants: K350CF and LT25, for the removal of COD and TSS from composting leachates. The use of poly ferric sulfate with a pH of 11 with 2 g/L of a coagulant, resulted in the highest removal rates with COD and

Table 7 Summary of physico-chemical treatment technologies' removal efficiencies for selected contaminants in composting leachates

Physico-chemical treatment		TSS		COD		BOD_5		TKN		NH_4		Authors
		Feed (mg/L)	Removal (%)	Feed (g/L)	Removal (%)	Feed (g/L)	Removal (%)	Feed (mg/L)	Removal (%)	Feed (mg/L)	Removal (%)	
1. Coagulation-flocculation												
Alum		–	–	22.3–45	18	–	–	–	–	–	–	Maleki et al. (2009)
Ferric chloride		–	–	22.3–45	28	–	–	–	–	–	–	Maleki et al. (2009)
Polyferric sulfate		15,200	51	124.5	49	–	–	–	–	–	–	Mahvi et al. (2015)
2. Electro-coagulation-flocculation												
Aluminium–Aluminium electrodes		–	99	14.5	96	–	–	–	–	–	–	Amin et al. (2014)
3. Membrane filtration												
Technology	Membrane											
UF/NF	Filmtec NF90	1820 ± 75	–	23.9 ± 1	97.3–98.7	–	–	1495 ± 50	–	754	78.7–85.2	Cakmakci and Ozyaka (2013)
coagulation/NF	Osmonics DL	–	–	31	99.6	–	–	–	–	–	–	Simonič (2017)
Lime precipitation/UF/RO	Osmonics AD	0.11–14,430	100	1.46–42.4	95	0.36–17.4	91	345–18,570	51.4	218–21,180	93	Gagnaire et al. (2012)
microfiltration/UF/RO	Filmtec BW30	–	–	23.9 ± 1	98.3	–	–	–	–	–	–	Seyda Özyaka et al. (2015)

TSS removals of 49% and 51%, respectively. On the other hand, Angadi et al. (2015) tested coagulation-flocculation for heavy metal removal (Angadi et al. 2015). The combination of alum (1 g/L) with polyelectrolyte (0.3 g/L) at pH 7 provided the highest performances with removals of 92.4%, 77.3% and 80.1% for cadmium, chromium, and nickel, respectively. Another study conducted by Maleki et al. (2009) also showed very efficient removals of heavy metals (Cd, Cr, Cu, Zn, Ni) (> 90%) despite low COD removal (18%) using alun (1.4 g/L) at a pH of 6.5 (Maleki et al. 2009). The treatment efficacies of each of these combinations of coagulant and flocculent are summarized in Table 7.

The use of coagulation-flocculation as a standalone process for composting leachate treatment has been well discussed in terms of removal efficiencies for COD, TSS and heavy metals. However, studies are lacking information regarding some important contaminants such as ammoniacal nitrogen and phosphorus. Furthermore, the production of sludge and the associated management cost have not been discussed.

3.3.2 Electro-coagulation/flotation

Electro-coagulation–flotation (ECF) is a physico-chemical treatment method that relies on the conduction of DC current in polluted wastewater and the dissolution of metallic electrodes (monopolar or bipolar connection) without the addition of chemical substances. Flotation is also provided, by means of aeration equipments and from the chemical reduction of water in gaseous hydrogen at the cathode, in order to enhance mixing and avoid concentration polarization (Amin et al. 2014). The advantages of ECF are: simplicity of design and operation, low retention time, low sludge production, lack of chemical addition, and fast sedimentation of flocs (Elyasi, Amani et al. 2015). However, the applications of EFC are limited to COD, phosphorus and TSS removal. Treatment efficacies of the ECF process are summarized in Table 7.

Amani et al. (2014) conducted a study on raw composting leachates in order to analyze and correlate the interactive effects of operational factors (such as influent COD, voltage, electrolysis time (ET), and electrode distance (ED)) on the efficiency of COD and TSS removal for various electrode configurations (Al–Al, Al–Fe, Fe–Al, Fe–Fe). Their results indicated that the best arrangement (Al–Al) provided the higest removal efficiencies (96% of COD and 99% of TSS) in the following conditions: voltage = 21 V, ET = 75 min, and ED = 3 cm (Amin et al. 2014; Amani et al. 2015).

Following this study, the same group of researchers conducted two studies in which ECF was used as a post-treatment for anaerobic biological treatments in order to meet discharge criteria. In both studies, the ECF reactor was set up to follow the previously identified ideal conditions. In the first study, conducted on the effluent of an anaerobic baffled reactor (ABR), the EFC post-treatment increased COD and TSS removal by an average of 8 and 9%, respectively (Elyasi et al. 2015). In the second study, conducted on the effluent of an expanded granular sludge bioreactor (EGSB), the ECF reactor increased COD and TSS removal up to 21 and 53%, respectively (Dastyar et al. 2015). In both cases, the ECF increased the quality of the effluent up to discharge standards, regarding both TSS and COD concentrations. However, ECF cannot be used as a standalone process due to its lack of efficiency for the treatment of nitrogen contaminants.

3.3.3 Filtration

Membrane filtration has been used for a large range of industrial applications (Roy et al. 2017), including composting leachate treatment. The filtration capacities (flux and removal) of membrane technologies are significantly affected by the molecular weight cut off and manufacturing materials of the membrane layers (Simonič 2017). The main advantages of membrane filtration are the commercial availability of a large range of membranes with a large range of molecular weight cut-offs (MWCO), and the production of a treated water (permeate) with a constant quality notwithstanding contaminant concentration variations in the feed. Despite the continuous improvement of membranes, high operating costs when compared to conventional biological treatments as well as membrane fouling remain the major drawbacks of membrane filtration technologies. Furthermore, the management of the concentrate can also be a major issue regarding the application.

In order to produce safely disposable wastewater from composting leachates, the use of nanofiltration (NF) and reverse osmosis (RO) have been studied and the separation efficiencies are summarized in Table 7. Using NF technology, Cakmakci and Ozyaka (2013) tested four membranes: the NP030 and NP010 (Microdyn Nadir), and the NF270 and NF90 (Dow-Filmtec). Simonic (2017) tested two other NF membranes: the DK and DL (Osmonics Desal). In order to prevent clogging of the membrane units with TSS, pretreatment was applied in both studies. Cakmakci & Ozyaka (2013) used a UC030 ultrafiltration membrane (MWCO of 30 kDa) and reported an initial COD removal of 17% and no significant removal of NH_4^+. Simonic (2017) used polyaluminium chloride (10 mL/L) as a coagulant agent and reported an initial COD and colloid removal of 75%. Then, the use of NF90 and DL membranes resulted in the highest COD removal efficiencies (98.7 and 99.6%, respectively). Cakmakci and Ozyaka (2013) also reported NH_4^+ removals between 79 and 85%, which correlates to the percentage of dissolved combined amino acids (Cakmakci and Ozyaka 2013; Simonič 2017).

Using RO technology, Gagnaire et al. (2012) tested the AD (Osmonics Desal), and Seyda Özyaka et al. (2015) tested both the BW30 and SW30 (Dow-Filmtec). As for NF, pretreatment was necessary in order to prevent the clogging of the membranes. Gagnaire et al. (2012) used chemical precipitation using lime, followed by an ultrafiltration (MWCO of 150 kDa). They reported an optimum lime dose of 384 mg N-NH_4/g $Ca(OH)_2$, which resulted in N-NH_4 and COD removals of 45–77 and 48%, respectively. The nanofiltration that followed increased COD removal by 8%, leading to total removals of 56 and $> 99.9\%$ for COD and TSS, respectively. Seyda Özyaka et al. (2015) used a sequence of micro and ultrafiltration membranes, namely the MP005 and ZW-UF, respectively. They reported 12% COD removal. Then, the use of BW30 and AD RO membranes resulted in COD removals of 98.3 and 95%, respectively. Again with the BW30 membrane, Gagnaire et al. (2012) also reported TSS, BOD_5 and N-NH_4^+ removals of 100, 91 and 93%, respectively (Gagnaire et al. 2012; Seyda Özyaka et al. 2015).

In terms of fouling, the direct use of RO membranes is not viable. An instantaneous loss of 80% in permeate flow is observed as soon as raw composting leachate filtration begins (Gagnaire et al. 2012). According to Simonic (2017), the main foulant material could be hydrophobic natural organic matter (NOM). They observed that NOM fouling layers lead to cake enhanced concentration polarization (Hoek and Elimelech 2003; Simonič 2017). Combined with an appropriate pre-treatment, NF and RO membranes represent a promising solution for COD, NH_4 and TSS removal in composting leachate.

3.4 Advanced technologies

Advanced oxidation processes (AOPs) differ from conventional oxidation process in their capacity of producing hydroxyl radicals (OH·), a highly reactive oxidative agent. Due to their high reactivity and non-selectivity, hydroxyl radicals are able to oxidize organic matter efficiently (Matilainen and Sillanpää 2010). Therefore, over the past few decades, the use of AOPs for the treatment of complex and refractory organic contaminants in wastewater gained in popularity (Wang et al. 2003). Due to the high organic content of composting leachates, the uses of AOPs as a primary treatment is not advantageous since the oxidant species consumption would be too high. Despite this consideration, an attempt at composting leachate treatment by Fenton reaction was performed by Trujillo et al. (2006). Under the optimal conditions for Fenton's reaction ($[Fe_2^+]/[COD] = 0.1$ with $[H_2O_2]/[COD] = 1$), they obtained COD and BOD5 removals of 75–77 and 90–98%, respectively (Trujillo et al. 2006).Other AOPs, such as O_3, O_3/persulfate, O_3/H_2O_2, and photocatalytic oxidation (UV/TiO_2), have been investigated in terms of their efficiency as a post-treatment step of composting leachate treated by biological processes.

3.4.1 *AOPs for the post-treatment of biologically-treated composting leachates*

Due to the refractory nature of composting leachates organic contamination, biological processes are often insufficient in obtaining an effluent that meets disposal criteria. In order to develop a sufficient treatment, AOPs have been studied for their ability to improve the removal of these hardly biodegradable organic molecules. The efficiencies of the studied AOPs in the post-treatment of biologically treated composting leachates are summarized in Table 8.

O_3 is a widely used oxidant in wastewater treatment. Mokhtarani et al. (2014) and Soubh and Mokhtarani (2016) studied the single ozonation process in the post-treatment of low COD (< 1 g/L) treated composting leachate. COD removals were similar in both studies (51 and 40%) (Mokhtarani et al.

Table 8 Efficiency of AOPs for the post-treatment of biologically-treated composting leachates

Advanced oxidation technology	Primary treatment	Contact time (h)	Optimum conditions	COD		BOD_5		BOD_5/COD		Authors
				Feed (g/L)	Removal (%)	Feed (g/L)	Removal (%)	Initial	Final	
1. Ozone										
O_3	Biological treatment	0.67	0.4 g O_3/h	0.5–1	51	<0.05	–	–	–	Mokhtarani et al. (2014)
H_2O_2/O_3	Anaerobic migrating blanket reactor	4.5	pH 8.5, 2.5 g/L H_2O_2, 1 g/h O_3	40.1 ± 7.1	72	22.2 ± 3.7	79	0.53	0.4	Amin and Ahmad Moazzam (2014)
O3/persulfate	Biological treatment	3.5	pH 9, 0.79 g O_3/h, 4500 mg/L $Na_2(SO_4)_2$	0.75	87	0.95	–	0.13	0.61	Soubh and Mokhtarani (2016)
2. Other AOPs										
Fenton's reaction	Activated sludge process	1.3	pH 4.5, 2 mL H_2O_2/L, 1.75 g/L $FeSO_4$	0.73 ± 0.1	56	0.15 ± 0.5	–	0.2	–	Mahdad et al. (2016)
Photo-Fenton's reaction	Activated sludge process	1.3	pH 4.5, 1 mL H_2O_2/L, 0.75 g/L $FeSO_4$	0.73 ± 0.1	70	0.15 ± 0.5	–	0.2	0.52	Mahdad et al. (2016)
UV/TiO_2	Anaerobic/aerobic reactor	21.5	pH 5.7, 7.5 mW/cm^2, 48.8 g TiO_2/m^2	0.45	58	<0.15	–	0.03	0.22	Mokhtarani et al. (2016)

2014, Soubh and Mokhtarani 2016). However, Soubh et al. reported a higher efficiency with an ozone consumption rate of 1.16 mg O_3/mg COD compared to the 2.8–4.1 mg O_3/mg COD rate obtained by Mokhtarani et al. (2014) despite the fact that they were using similar reactors. In order to improve the COD removal, Soubh and Mokhtarani (2016) also tested the combination of O_3/persulfate(Soubh and Mokhtarani 2016). The addition of 4500 mg/L of persulfate increased the COD removal rate from 40 to 87%, and increased the BOD_5/COD from 0.13 to 0.61. The combination of O_3/H_2O_2 has also been tested as a post-treatment for high strength composting leachates (COD $= 40.1 \pm 7.1$ mg/L). Despite high initial concentrations, the O_3/H_2O_2 process reduced the COD and BOD_5 concentrations by 72 and 79%, respectively(Amin and Ahmad Moazzam 2014). However, compared to the O_3/persulfate process, the O_3/H_2O_2 process reduced the BOD_5/COD from 0.53 to 0.4, indicating that biodegradable organic matter is more likely to be oxidized when using H_2O_2. This observation is in accordance with the findings of Trujillo et al. (2006).

A conventional Fenton process and a photo-Fenton process (combination of Fenton's reagent with UV light to further increase hydroxyl radical production) were compared as post-treatment. With regard to Fenton oxidation (pH of 4.5, 1.75 g/L $FeSO_4$, and 2 ml/L H_2O_2), 55.9% COD and 65.7% colour were removed. With regard to photo-Fenton (pH of 4.5, 0.75 g/L $FeSO_4$, and 1 ml/H_2O_2), 73.8% COD and 83.6% colour were removed. Results also indicated that the biodegradability of the effluent increased with the photo-Fenton process (BOD_5/COD of 0.52 compared to 0.2) (Mahdad et al. 2016).

The emerging UV/TiO_2 photocatalytic oxidation process was also investigated as a post-treatment. This process generates in situ hydroxyl radicals under ambient conditions. When the TiO_2 catalyst is exposed to UV radiation, an electron (e^-)—hole (h^+) pair is produced. In most cases, on the surface of the catalyst, h^+ reacts easily with surface-bound H_2O to produce OH• free radicals and H^+ (Mokhtarani et al. 2016). In addition to being the first to report the use UV/TiO_2 in composting leachate post-treatment, Mokhtarani et al. (2015) also chose to immobilize the TiO_2 nanoparticles on a concrete surface. After 21.5 h of radiation with 7.5 mW/cm^2 light intensity, the maximum COD and colour removal achieved were 58 and 36%, respectively. The biodegradability of the effluent also increased (BOD_5/COD of 0.22 compared to 0.03) (Mokhtarani et al. 2016).

AOPs are efficient technologies to use in the removal of refractory organic contamination (BOD_5/COD < 0.2) found in composting leachates. Since they are more largely effective for COD and colour removal, AOPs should be considered as a biologically-treated effluent post-treatment. However, the addition of a physico-chemical technology should still be considered in order to remove TSS.

3.5 Composting leachate reuse and transformation

Composting leachate treatment comes with costs of operation, labor, and time. To transform these costs into profit, studies have proposed solutions that create added-value products or reuse composting leachate to replace commercial products.

Justin et al. (2010) compared the impact of landfill versus composting leachates on the growth of Salix species and Populus. The beneficial attributes of these tree species include: an appropriate phytoremediation capacity, a tolerance for extreme growing conditions, and a high-yield of biomass production. This biomass can, for instance, be transformed for fuel, chipboard, paper, and charcoal production. Despite the high nutrient content (up to 1105 mg/L N, 372 mg/L P and 2888 mg/L K) of composting leachate, there seemed to be less potential for its use due to its high concentrations variability (Justin et al. 2010). Over the course of the same year, Zhou et al. (2010) conducted a similar study using composting leachates to irrigate Impatiens, a common garden flower (Zhou et al. 2010). However, instead of using raw composting leachates, they added a microbial culture to the leachate and stored the mixture for nine weeks. This preconditioning reduced the COD and BOD_5 concentration by 9.6 and 6.7% respectively. Then, they diluted the conditioned leachate (1:10) before the irrigation in order to balance the nutrients concentrations. In contrast to the results of Justin et al. (2010), the use of pretreated composting leachates promoted the growth of Impatiens. Other composting leachate preconditioning techniques to reduce organic contamination have also been studied, such as the use of bioreactors with various HRT. Romero et al. (2013) used an anaerobic continuous stirred tank reactor (A-CSTR) to reduce the COD of the raw leachate (87%

removal) (Romero et al. 2013). While no significant effect was observed on the germination index, the biological treatment reduced odour.

In order to transform composting leachates contamination in a value-added product, Liu et al. (2015) evaluated the possibility of fermentative hydrogen (H_2) production using leachates as an alternative source of nutrients (Liu et al. 2015). To compensate for the lack of phosphorus required for fermentative H_2 production, orthophosphate and pyrophosphate were separately added into leachate. With COD/P ratio of 27.6/1 at pH of 5, the H_2 yield obtained was 1.95 mol H_2/mol glucose. These results are promising for future usage of fresh leachate and phosphate contained waste for biohydrogen production.

The transformation and reuse of composting leachates would bring environmental benefits through the planting of green areas, which would in turn help contain and limit erosion while also increasing carbon sequestration. Furthermore, compost leachates can replace chemical fertilizer in organic cultivation farms (Kim et al. 2015). Nevertheless, these alternative approaches have certain obstacles to overcome before becoming technically feasible. Major issues include the presence of: toxic organic compounds with low molecular weights (Romero et al. 2013), high concentrations of COD and BOD_5, phytotoxic substances like and ammonia, and high concentrations of TDS. Furthermore, the presence of heavy metals can cause partial damage to plant leaves, reduce their chlorophyll production, and can be bioacculumated in the plants.

3.6 Operation costs of leachate treatment technologies

Operating cost is a major factor when selecting the appropriate leachate treatment technology. Among all the technology studied for the treatment of composting leachates, biological treatment appears to be the most economical option with operating costs ranging from 0.20 to 3.20 USD/m^3 (Robinson 2005; Kurniawan et al. 2006; Cassano et al. 2011). Robinson (2005) conducted a comprehensive analysis on MBR's operating costs and found that MBR can treat leachates with costs ranging from 0.20 to 0.75 USD/m^3 (Robinson 2005). Membrane filtration is also a technology of choice for leachate treatment. As a standalone treatment, RO has operating costs ranging from 3.80 to 10 USD/m^3 (Thörneby et al. 2003; Kurniawan et al. 2006). Combined with a pretreatment such as UF membranes, the operating cost is reduced from 0.80 to 1.05 USD/m^3 (Ozturk et al. 2003). However, this cost does not take into account concentrate management, which can be prohibitive for leachates with a high dissolved solids content.

The operating costs of physico-chemical treatments for ammoniacal nitrogen, such as air stripping and struvite precipitation, is 0.52 and 2–4.45 USD/m^3, respectively (Ozturk et al. 2003; Kurniawan et al. 2006). Despite having similar operating costs as biological treatment, these technologies are only effective on one contaminant, and they also have to be incorporated in a sequence of technologies. Thus, the entire treatment can quickly become costly. A similar analysis can be done for coagulation coupled with activated carbon. The operating cost of this technology is approximately 2.30 USD/m^3, though it only removes soluble organic contaminants and suspended solids (Kurniawan et al. 2006). RO coupled with NF has similar operating costs, and therefore seems to be a better economical and technical option in terms of physico-chemical treatment technology.

Similarly to physico-chemical technologies, advanced oxidation technologies cannot be used standalone due to the high concentration of organic pollutants in the composting leachates. The operating cost of O_3 used as a post-treatment ranges from 0.90 to 4 USD/m^3 (Tizaoui et al. 2007; Cassano et al. 2011) and its combination with H_2O_2 (2 g/L) costs approximately 3.05 USD/m^3 (Tizaoui et al. 2007). The Fenton's process as a standalone technology is even more costly, with operating costs ranging from 1.30 to 17.10 USD/m^3 (Singh et al. 2013). A potentially economical option to consider is the use of the Fenton's process as a pre-treatment to improve the biodegradability of the organic contaminants before of a biological treatment. Xie et al. (2010) estimated the operating cost of the combination of the Fenton's process with a biofilter at 2.10 USD/m3, which is in the range of costs of biotreatment technologies (Xie et al. 2010).

Finally, thermal technology operating costs have also been assessed. With operating costs ranging from 30 to 70 USD/m^3 (evaporation and thermal oxidation), these technologies should only be considered as a last resort, despite their very high removal efficacy for almost all contaminants (> 99% removal) (Ozturk et al. 2003).

4 Summary

The growing interest in the composting of organic waste will inevitably lead to an increase in composting leachate production. Composting leachate contamination is characterized by high concentrations of moderately biodegradable organic matter and nutrients, and by the presence of heavy metals and plasticizers, all of which exceed NRWQC criteria for freshwater aquatic life. Furthermore, concentrations of contaminants are highly variable due to two main factors: (1) the type of composted organic wastes (feedstock) and (2) the type of composting technology. The heterogeneous composition and seasonal variability of the feedstock both influence the composting leachate composition. The composting technology, on the other hand, influences the amount of water involved in the process and the oxic conditions inside the organic wastes being composted. The main contaminants of composting leachates are summarized and described in Table 9.

Despite the fact that the types of contaminants found in composting leachates are quite common, selecting and designing a water treatment technology capable of producing a safely disposable treated effluent poses a challenge due to the wide range of contaminant types and concentrations. To date, the treatment of composting leachate with biological (biofilters, anaerobic bioreactors and MBR), physico-chemical (coagulation-flocculation, electro-coagulation/flotation, nanofiltration and reverse osmosis), and advanced oxidation technologies (Fenton's reaction, O_3, O_3/H_2O_2, O_3/persulfate and UV/TiO_2) have been reported in the literature.

Biological processes are the most financially sound solutions. MBRs operating costs are the lowest, ranging from 0.21 to 0.75 USD/m^3. The most economical physico-chemical treatment option seems to be the combination of NF and RO (0.80–1.05 USD/m3). However, the concentrate management cost can be prohibitive for leachates with high TDS concentrations. Finally, oxidation processes as standalone technologies are the most costly to operate (1.30–17.10 \$/m^3). However, combing them with biological processes as a pre- or post-treatment has the potential of being economical. In the present context, none of these technologies can produce an effluent meeting all NRWQC criteria, which is a major concern considering the increasing number of composting facilities. Thus far, MBRs and membrane filtration technologies are the most promising possibilities. Unlike the other proposed technologies, these two are efficient for the simultaneous removal of COD, TSS and NH_4.

Table 9 Composting leachates contaminants properties

Organic matter	
Moderately biodegradable	$BOD_5/COD \pm 0.33$
High molecular weight	1000–3000 kDa
Highly aromatic	Fulvic- and humic-like substances
Presence of volatile fatty acids (VFAs)	
Presence of toxic organic contaminants	DEHP, BPA
Nitrogen	
Proteinaceous and amines compounds	75–85%
Ammonium salts	15–25%
Highly biodegradable	$N\text{-}NH_4/NTK > 0.5$
Integrated to fulvic- and humic-like substances	
Heavy metals	
Common heavy metals	Cd, Cu, Ni, Pb and Zn
Complexation properties	Strong complexation capability with fulvic- and humic like substance
Solubility	High, due to complexation

5 Conclusion

To face the growing production of composting leachates and provide an integrated solution for the treatment of this particular wastewater, further researches are required. In the light of the information presented in this review paper, future research perspectives in the field of composting leachate treatment are the following.

1. The presence of toxic organic pollutants in composting leachates have been reported by Fromme et al. (2002) and Marttinen et al. (2004). However, no subsequent studies conducted on the treatment of composting leachates have assessed the presence of TOPs or the capacity of the different technologies to treat these contaminants. Thus, in the future, the fate of TOPs during the leachate treatment should be considered.
2. As mentioned in Sect. 3.1, there is a significant lack of information disclosure regarding the use of MBR for composting leachate treatment. In order to fill this information gap, composting leachate contaminant removal kinetics in aerobic and anaerobic MBR should be studied. This information is required in order to design reactors with suitable operating conditions and optimal removal efficiencies. Furthermore, economical and technical comparison with conventional technologies (such as aerated lagoons or activated sludge) should be performed when assessing the efficiency of more complex biological technologies.
3. The operation of a MBR based on the principle of UCT-type processes for the treatment of composting leachate should be studied. UCT-type processes consist of a series of anaerobic, anoxic, and aerobic biological reactors. The combination of these oxic conditions provides significant advantages compared to other biological treatment technologies, namely, the enhanced COD and phosphorus removal efficiencies, and nitrification and denitrification reaction completions over the course a single process (Kuba et al. 1997). Considering the reported treatment efficiencies of single tank MBR for the treatment of composting leachates, the addition of a UCT-type process could produce an effluent that meets the NRWQC criteria without further treatment.
4. Along with the development of a UCT-MBR, other process combinations should be considered. The combination of biological treatment with advanced oxidation process pre- and post-treatment is a financially and technically attractive solution. Biological treatment processes have the ability to reduce the nitrogen and biodegradable organic contamination at low cost, while the AOPs are able to oxidize refractory organic compounds and TOPs. A promising combination that should be analyzed for further development is the MBR-AOP. In addition to the submerged membrane of the MBR completely removing the TSS, the bioreactor could be operated with low HRT since any untreated COD would be removed by the AOP post-treatment.
5. Further research should be conducted in the use of composting leachates as a culture media in the production of value-added molecules with high market values. This solution would increase the profitability of composting facilities, making them an even more feasible waste management solution.

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