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Review

Landfill leachate treatment: Review and opportunity

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Abstract

In most countries, sanitary landfilling is nowadays the most common way to eliminate municipal solid wastes (MSW). In spite of many advantages, generation of heavily polluted leachates, presenting significant variations in both volumetric flow and chemical composition, constitutes a major drawback. Year after year, the recognition of landfill leachate impact on environment has forced authorities to fix more and more stringent requirements for pollution control. This paper is a review of landfill leachate treatments. After the state of art, a discussion put in light an opportunity and some results of the treatment process performances are given. Advantages and drawbacks of the various treatments are discussed under the items: (a) leachate transfer, (b) biodegradation, (c) chemical and physical methods and (d) membrane processes. Several tables permit to review and summarize each treatment efficiency depending on operating conditions. Finally, considering the hardening of the standards of rejection, conventional landfill leachate treatment plants appear under-dimensioned or do not allow to reach the specifications required by the legislator. So that, new technologies or conventional ones improvements have been developed and tried to be financially attractive. Today, the use of membrane technologies, more especially reverse osmosis (RO), either as a main step in a landfill leachate treatment chain or as single post-treatment step has shown to be an indispensable means of achieving purification.

Keywords: Landfill leachate; Wastewater treatment; Review

Contents

Abbreviations: AOP, advanced oxidation processes; AS, activated sludge; BOD, biological oxygen demand; COD, chemical oxygen demand; DOC, dissolved organic carbon; GAC, granular activated carbon; HRT, hydraulic retention time; MAP, magnesium ammonium phosphate; MBBR, moving-bed biofilm reactor; MSW, municipal solid waste; MSWLF, municipal solid waste landfill; PAC, powdered activated carbon; RDVPF, rotary drum vacuum precoat filter; RO, reverse osmosis; SBR, sequencing batch reactor; SCBR, suspended-carrier biofilm reactor; SRT, sludge retention time; SS, suspended solids; TKN, total Kjeldahl nitrogen; TOC, total organic carbon; UASB, up-flow anaerobic sludge blanket; VFA, volatil fatty acids.

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1. Introduction

Increasingly affluent lifestyles, continuing industrial and commercial growth in many countries around the world in the past decade has been accompanied by rapid increases in both the municipal and industrial solid waste production. Municipal solid waste (MSW) generation continues to grow both in per capita and overall terms. For example, in 1997, waste production in Rio de Janeiro, Brazil, was 8042 tonnes day^{-1} compared to 6200 tonnes day^{-1} in 1994, despite the fact that population growth during that period was practically zero. Waste production increased by 3% and 4.5% per year between 1992 and 1996, respectively, in Norway and in the USA. During the latter part of the 1990s, annual waste production ranged from 300 to 800 kg per person in the more developed countries to less than 200 kg in other countries [1]. In 2002, French population produced 24 million of MSW, namely 391 kg per person [2].

The sanitary landfill method for the ultimate disposal of solid waste material continues to be widely accepted and used due to its economic advantages. Comparative studies of the various possible means of eliminating solid urban waste (landfilling, incineration, composting, . . ., etc.) have shown that the cheapest, in term of exploitation and capital costs, is landfilling. In 2002, 52% of waste production in France was landfilled into regulated centers [2]. Besides its economic advantages, landfilling minimizes environmental insults and other inconveniences, and allows waste to decompose under controlled conditions until its eventual transformation into relatively inert, stabilized material.

So, the worldwide trend is for controlled sanitary landfilling as the preferred means of disposing of both solid urban refuse and a large proportion of solid industrial waste. It concerns both industrialized cities (11,500 tonnes day^{-1} of MSW in Mexico city) and rural areas (about 40,000 tonnes $year^{-1}$ in the Kyletalesha landfill site, Ireland). Also, recent estimates indicates that 52, 90 and 95% of urban wastes are disposed of at landfill sites, respectively, in Korea, Poland and Taiwan. However, the release from a sanitary landfill consist mainly of leachate which has became the subject of recent interest as a strongly polluted wastewater and biogas, that is a resource which can be utilized for energy production [3].

There is now extensive scientific literature on the collection, storage and suitable treatment of its highly contaminated leachates, threatening surface and ground waters. Fig. 1 summarizes the evolution of main published research, concerning landfill leachate treatment, reported in the world's journal and patent literature since 1973 (data extracted from Chemical Abstracts).

Leachates are defined as the aqueous effluent generated as a consequence of rainwater percolation through wastes, biochemical processes in waste's cells and the inherent water content of wastes themselves. Leachates may contain large amounts of organic matter (biodegradable, but also refractory to biodegradation), where humic-type constituents consist an important group, as well as ammonia-nitrogen, heavy metals, chlorinated organic and inorganic salts. The removal of organic material based on chemical oxygen demand (COD), biological oxygen demand (BOD) and ammonium from leachate is the usual prerequisite before discharging the leachates into natural waters. Toxicity analysis carried out using various test organisms (Vibrio fisheri, Daphnia similes, Artemia salina, Brachydanio rerio . . .) have confirmed the potential dangers of landfill leachates [4–8] and the necessity to treat it so as to meet the standards for discharge in receiving waters.

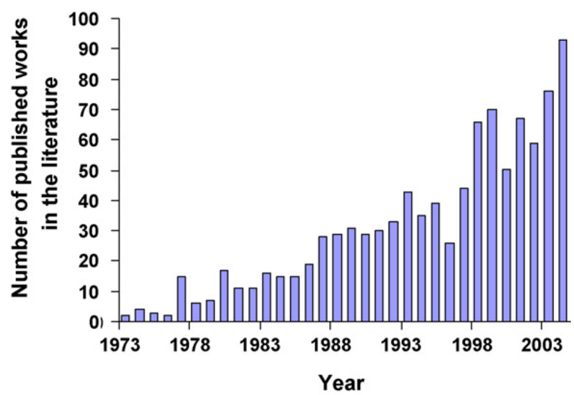


Fig. 1. Evolution of published works concerning landfill leachate treatment since 1973 (*source*: Chemical Abstracts).

According to this fact, governments apply enhanced regulation for non-biodegradable organic matter and for nitrogenous compounds. In 1997, French authorities have fixed more stringent requirements concerning discharge into surface waters (Table 1). Fortunately, the remarkable growth in economics and living standard has accelerated the development of water and wastewater purification technologies.

Table 1
Revised French regulation criteria (selected), in 1997

Item	Volumetric classification (kg day^{-1})	Criterion after revision (mg L^{-1})
COD	<100	300
	>100	125
TOC	–	70
Total suspended solids (TSS)	<15	100
	>15	35
BOD_5	<30	100
	>30	30
Total nitrogen	>50	30

In summary, MSW management constitutes today a major environmental, economical and social problem worldwide, mainly because the waste volume is growing faster than the world's population. Moreover, as stricter environmental requirements are continuously imposed regarding ground and surface waters, the treatment of landfill leachate becomes a major environmental concern. This review, therefore, focuses on the state of art in landfill leachate treatment and provides a comparative evaluation of various treatment processes. New treatment alternatives and conventional technology improvements are highlighted and examinated.

2. Leachate characteristics

The two factors characterizing a liquid effluent are the volumetric flow rate and the composition which in the case of leachate are related. Fig. 2 illustrates water cycle in a landfill. Leachate flow rate (*E*) is closely linked to precipitation (*P*), surface run-off (Rin, Rext), and infiltration (*I*) or intrusion of groundwater percolating through the landfill. Landfilling technique (waterproof covers, liner requirements such as clay, geotextiles and/or plastics) remains primordial to control the quantity of water entering the tip and so, to reduce the threat pollution [10]. The climate has also a great influence on leachate production because it affects the input of precipitation (*P*) and losses through evaporation (EV). Finally, leachates production depends on the nature of the waste itself, namely its water content and its degree of compaction into the tip. The production is generally greater whenever the waste is less compacted, since compaction reduces the filtration rate [10].

There are many factors affecting the quality of such leachates, i.e., age, precipitation, seasonal weather variation, waste type and composition (depending on the standard of living of the surrounding population, structure of the tip). In particular, the composition of landfill leachates varies greatly depending on the age of the landfill [11]. Fig. 3 [10] proposes anaerobic degradation scheme for the organic material in a sanitary landfill. In young landfills, containing large amounts of biodegradable organic matter, a rapid anaerobic fermentation takes place, resulting in volatile fatty acids (VFA) as the main fermentation products [12]. Acid fermentation is enhanced by a high moisture content or water content in the solid waste [13]. This early phase of a landfill's lifetime is called the acidogenic phase, and leads to the release of large quantities of free VFA, as much as 95% of the organic content [14]. As a landfill matures, the methanogenic phase occurs. Methanogenic microorganisms develop in the waste, and the VFA are converted to biogas (CH_4, CO_2). The organic fraction in the leachate becomes dominated by refractory (non-biodegradable) compounds such as humic substances [15].

The characteristics of the landfill leachate can usually be represented by the basic parameters COD, BOD, the ratio BOD/COD, pH, suspended solids (SS), ammonium nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN) and heavy metals. The leachate composition from different sanitary landfills, as reported in the literature, show a wide variation. Tables 2 and 3 summarize the ranges of leachate composition. These data show that the age of the landfill and thus the degree of solid waste stabilization has a significant effect on water characteristics. Values of COD vary from 70,900 mg L^{-1} with leachate sample obtained from the Thessaloniki Greater Area (Greece) to 100 mg L^{-1} with sample from an more than 10-year old landfill near Marseille (France). With few exceptions, the pH of leachates lie in the range 5.8–8.5, which is due to the biological activity inside the tip. It is also important to notice that the majority of TKN is ammonia, which can range from 0.2 to 13,000 mg L^{-1} of N. The ratio of BOD/COD, from 0.70 to 0.04, decrease rapidly with the aging of the landfills [15]. This is due to the release of the large recalcitrant organic molecules from the solid wastes. Consequently, old landfill leachate is characterized by its low ratio of BOD/COD and fairly high NH_3-N.

Although leachate composition may vary widely within the successive aerobic, acetogenic, methanogenic, stabilization stages of the waste evolution, three types of leachates have been defined according to landfill age (Table 4). The existing relation between the age of the landfill and the organic matter composition may provide a useful criteria to choose a suited treatment

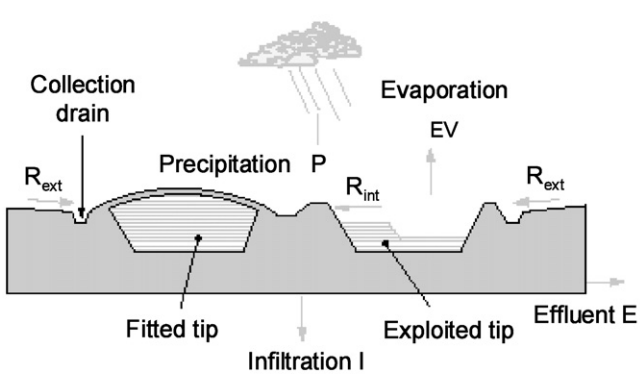


Fig. 2. Water cycle in a sanitary landfill [9].

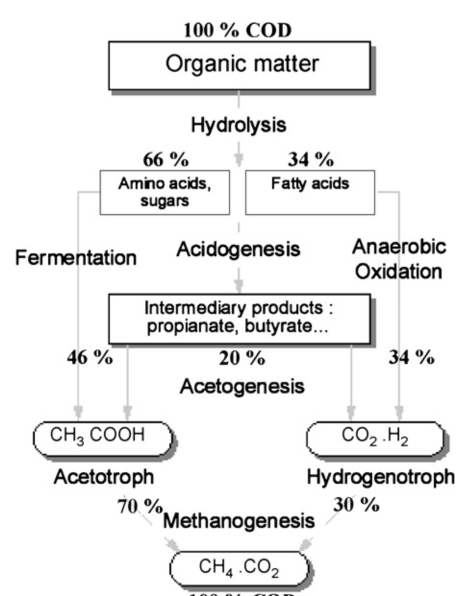


Fig. 3. COD balance of the organic fraction in a sanitary landfill [10].

Table 2
Leachate composition (COD, BOD, BOD/COD, pH, SS, TKN, NH_3-N)

Age	Landfill site	COD	BOD	BOD/COD	pH	SS	TKN	NH_3-N	Reference
Y	Canada	13,800	9660	0.70	5.8	–	212	42	[16]
Y	Canada	1870	90	0.05	6.58	–	75	10	
Y	China, Hong Kong	15,700	4200	0.27	7.7	–	–	2,260	[17]
Y	China, Hong Kong	17,000	7300	0.43	7.0–8.3	>5000	3,200	3,000	[18]
Y		13,000	5000	0.38	6.8–9.1	2000	11,000	11,000	
Y		50,000	22,000	0.44	7.8–9.0	2000	13,000	13,000	
Y	China, Mainland	1900–3180	3700–8890	0.36–0.51	7.4–8.5	–	–	630–1,800	[19]
Y	Greece	70,900	26,800	0.38	6.2	950	3,400	3,100	[20]
Y	Italy	19,900	4000	0.20	8	–	–	3,917	[3]
Y	Italy	10,540	2300	0.22	8.2	1666	–	5,210	[21]
Y	South Korea	24,400	10,800	0.44	7.3	2400	1,766	1,682	[22]
Y	Turkey	16,200–20,000	10,800–11,000	0.55–0.67	7.3–7.8	–	–	1,120–2,500	[23]
		35,000–50,000	21,000–25,000	0.5–0.6	5.6–7.0	–	–	2,020	
Y	Turkey	35,000–50,000	21,000–25,000	0.5–0.6	5.6–7.0	2630–3930	2,370	2,020	[24]
Y	Turkey	10,750–18,420	6380–9660	0.52–0.59	7.7–8.2	1013–1540	–	1,946–2,002	[25]
MA	Canada	3210–9190	–	–	6.9–9.0	–	–	–	[26]
MA	China	5800	430	0.07	7.6	–	–	–	[27]
MA	China, Hong Kong	7439	1436	0.19	8.22	784	–	–	[28]
MA	Germany	3180	1060	0.33	–	–	1,135	884	[29]
MA	Germany	4000	800	0.20	–	–	–	800	[30]
MA	Greece	5350	1050	0.20	7.9	480	1,100	940	[20]
MA	Italy	5050	1270	0.25	8.38	–	1,670	1,330	[31]
MA	Italy	3840	1200	0.31	8	–	–	–	[32]
MA	Poland	1180	331	0.28	8	–	–	743	[33]
MA	Taiwan	6500	500	0.08	8.1	–	–	5,500	[34]
MA	Turkey	9500	–	–	8.15	–	1,450	1,270	[35]
O	Brazil	3460	150	0.04	8.2	–	–	800	[7]
O	Estonia	2170	800	0.37	11.5	–	–	–	[36]
O	Finland	556	62	0.11	–	–	192	159	[37]
O	Finland	340–920	84	0.09–0.25	7.1–7.6	–	–	330–560	[5]
O	France	500	7.1	0.01	7.5	130	540	430	[38]
O	France	100	3	0.03	7.7	13–1480	5–960	0.2	[39]
O	France	1930	–	–	7	–	–	295	[40]
O	Malaysia	1533–2580	48–105	0.03–0.04	7.5–9.4	159–233	–	–	[41]
O	South Korea	1409	62	0.04	8.57	404	141	1,522	[42]
O	Turkey	10,000	–	–	8.6	1600	1,680	1,590	[43]

Y: young; MA: medium age; O: old; all values except pH and BOD/COD are in mg L^{-1}.

Table 3
Heavy metals composition in landfill leachate

Age	Landfill site	Fe	Mn	Ba	Cu	Al	Si	Reference
Y	Italy	2.7	0.04	–	–	–	–	[21]
MA	Canada	1.28–4.90	0.028–1.541	0.006–0.164	–	<0.02–0.92	3.72–10.48	[26]
MA	Hong Kong	3.811	0.182	–	0.12	–	–	[28]
MA	South Korea	76	16.4	–	0.78	–	–	[22]
MA	Spain	7.45	0.17	–	0.26	–	–	[44]
O	Brazil	5.5	0.2	–	0.08	<1	–	[7]
O	France	26	0.13	0.15	0.005–0.04	2	<5	[39]
O	Malaysia	4.1–19.5	15.5	–	–	–	–	[41]
O	South Korea	–	0.298	–	0.031	–	–	[42]

Y: young; MA: medium age; O: old; all values are in mg L^{-1}.

process. The aim of this article is to propose a comprehensive review of landfill leachate treatment processes and to understand their evolution with the increasingly stringent discharge standards on last decades. To evaluate their treatment performances on the basis of COD, NH_3-N and heavy metal, selected information on pH, dose required, strength of wastewater in terms of COD, NH_3-N and heavy metal concentration, as well as treatment efficiency is presented.

3. Review and evolution of landfill leachate treatments

3.1. Conventional treatments

Conventional landfill leachate treatments can be classified into three major groups: (a) leachate transfer: recycling and combined treatment with domestic sewage, (b) biodegradation: aerobic and anaerobic processes and (c) chemical and physical methods: chemical oxidation, adsorption, chemical precipitation, coagulation/flocculation, sedimentation/flotation and air stripping.

3.1.1. Leachate transfer

3.1.1.1. Combined treatment with domestic sewage. Few years ago, a common solution was to treat the leachate together with municipal sewage in the municipal sewage treatment plant. It was preferred for its easy maintenance and low operating costs [45]. However, this option has been increasingly questioned due to the presence in the leachate of organic inhibitory compounds with low biodegradability and heavy metals that may reduce treatment efficiency and increase the effluent concentrations [25]. An argument in favour of this alternative treatment is that nitrogen (brought by leachate) and phosphorus (brought by sewage) do not need to be added at the plant. Among the few studies published, authors tried to optimise the volumetric ratio of leachate in the total wastewater. Combined treatment was investigated by Diamadopoulos et al. [46] using a sequencing batch reactor (SBR) consisting of filling, anoxic, oxic and settling phases. When the ratio of sewage to leachate was 9/1, nearly 95% BOD and 50% nitrogen removals were obtained at the end of the daily cycles. COD and NH_4^+-N reduction decreased with increasing landfill leachate/domestic wastewater ratio [47]. Moreover, the effluent quality may be improved with powdered activated carbon (PAC) addition, particularly if the leachate input exceeds 10%. Other researchers (Table 5) studied the co-treatment of leachate and sewage [10,48,49] and showed similar results.

3.1.1.2. Recycling. Recycling leachate back through the tip has been largely used in the past decade because it was one of the least expensive options available [10]. Recently, authors showed benefits of this technique. Bae et al. [50] reported that leachate recirculation increased the moisture content in a controlled reactor system and provided the distribution of nutrients and enzymes between methanogens and solid/liquids. Significant lowering in methane production and COD was observed when the recirculated leachate volume was 30% of the initial waste bed volume [51]. Also, Rodriguez et al. [52] reported a 63–70% COD lowering in an anaerobic pilot plant with recirculation. The leachate recycle not only improves the leachate quality, but also shortens the time required for stabilization from several decades to 2–3 years [53]. Although positive effects have been reported on solid waste degradation, limited data are available (Table 6) concerning the recirculation rate impact on treatment efficiency in controlled anaerobic digesters [52,54,55]. High recircula-

Table 4
Landfill leachate classification vs. age [15]

	Recent	Intermediate	Old
Age (years)	<5	5–10	>10
pH	6.5	6.5–7.5	>7.5
COD (mg L^{-1})	>10,000	4000–10,000	<4000
BOD_5/COD	>0.3	0.1–0.3	<0.1
Organic compounds	80% volatile fat acids (VFA)	5–30% VFA + humic and fulvic acids	Humic and fulvic acids
Heavy metals	Low–medium		Low
Biodegradability	Important	Medium	Low

Table 5
Combined treatment with domestic sewage

Feeding				Operational conditions					Performance removal (%)	Reference
COD (mg L^{-1})	BOD/COD	pH	From	Kind of reactor	Volume of reactor (L)	T (°C)	HRT (days)	Volumetric ratio (%)		
1090	0.4	–	Landfill leachate + municipal sewage	SBR	–	20	1	0.11 (1:9)	95 BOD	[46]
10,750	0.59	8.2	Landfill	AS	2	–	1.3	6.7–13.3	60–90 COD	[47]
2431–37,024	0.2–0.4	7.3–7.9	Landfill	AS	2	22	1–10	5–20	16–88 COD	[49]
10,750–18,420	0.55	7.7–8.2	Landfill leachate + municipal sewage	AS	3.6 (aeration tank); 2.5 (settling tank)	–	–	5–25	–	[25]

tion rates may adversely affect anaerobic degradation of solid wastes. For instance, Ledakowicz and Kaczorek [57] observed that leachate recirculation can lead to the inhibition of methanogenesis as it may cause high concentrations of organic acids (pH < 5) which are toxic for the methanogens. Furthermore, if the volume of leachate recirculated is very high, problems such as saturation, ponding and acidic conditions may occur [58,59].

3.1.2. Biological treatment

Due to its reliability, simplicity and high cost-effectiveness, biological treatment (suspended/attached growth) is commonly used for the removal of the bulk of leachate containing high concentrations of BOD. Biodegradation is carried out by microorganisms, which can degrade organics compounds to carbon dioxide and sludge under aerobic conditions and to biogas (a mixture comprising chiefly CO_2 and CH_4) under anaerobic conditions [10]. Biological processes have been shown to be very effective in removing organic and nitrogenous matter from immature leachates when the BOD/COD ratio has a high value (>0.5). With time, the major presence of refractory compounds (mainly humic and fulvic acids) tends to limit process's effectiveness.

3.1.2.1. Aerobic treatment. An aerobic treatment should allow a partial abatement of biodegradable organic pollutants and should also achieve the ammonium nitrogen nitrification. Aerobic biological processes based on suspended-growth biomass, such as aerated lagoons, conventional activated sludge processes and sequencing batch reactors (SBR), have been widely studied and adopted [28,60–63]. Attached-growth systems have recently attracted major interest: the moving-bed biofilm reactor (MBBR) and biofilters. The combination of membrane separation technology and aerobic bioreactors, most commonly called membrane bioreactor, has also led to a new focus on leachate treatment.

3.1.2.1.1. Suspended-growth biomass processes.

Lagooning. Aerated lagoons have generally been viewed as an effective and low-cost method for removing pathogens, organic and inorganic matters. Their low operation and maintenance costs have made them a popular choice for wastewater treatment, particularly in developing countries since there is a little need for specialised skills to run the system [64]. Wide variations in the standard performance of lagoon systems have been reviewed in the literature (Table 7). Maehlum [66] used onsite anaerobic–aerobic lagoons and constructed wetlands for biological treatment of landfill leachate. Overall N, P and Fe removals obtained in this system were above 70% for diluted leachate. Orupold et al. [36] studied the feasibility of lagooning to treat phenolic compounds as well as organic matter. Abatement of 55–64% of COD and 80–88% of phenol was achieved. However, as stricter requirements are imposed, lagooning may not be a completely satisfactory treatment option for leachate in spite of its lower costs [68]. In particular, authors claimed that the temperature dependence of lagooning is a significant limitation because it mainly affects microbial activity.

Table 6
Landfill leachate recycling

Feeding			Operational conditions			Performance removal (%)	Reference
COD (mg L^{-1})	pH	From	Volume of reactor (L)	T (°C)	Recirculation rate (L day^{-1})		
80,000	5.5–6.5	Pilot plant	707	36	–	98 COD	[56]
47,000–52,000	–	Pilot plant	70	35	9–21	–	[55]
716–1765	7.58–7.60	Pilot plant	–	–	40	63–70 COD	[52]
2560–5108	8.00–8.43	Landfill	–	–	40		

Activated sludge processes. They are extensively applied for the treatment of domestic wastewater or for the co-treatment of leachate and sewage. However, this method has been shown in the more recent decades to be inadequate for handling landfill leachate treatment [69]. Even if processes were proved to be effective for the removal of organic carbon, nutrients and ammonia content, too much disadvantages tend to focus on others technologies:

- inadequate sludge settleability and the need for longer aeration times [70],
- high energy demand and excess sludge production [37],
- microbial inhibition due to high ammonium-nitrogen strength [10].

Consequently, only few works are recently available concerning landfill leachate treatment by activated sludge methods (Table 8). Hoilijoki et al. [37] investigated nitrification of anaerobically pre-treated municipal landfill leachate in lab-scale activated sludge reactor, at different temperatures (5–10 °C) and with the addition of plastic carrier material. Aerobic post-treatment produced effluent with 150–500 mg COD L^{-1}, less than 7 mg BOD L^{-1} and on an average, less than 13 mg NH_4^+-N L^{-1}. Addition of PAC to activated sludge reactors enhanced nitrification efficiency on biological treatment of landfill leachate [91].

Sequencing batch reactor. This system is ideally suited to nitrification–denitrification processes since it provides an operation regime compatible with concurrent organic carbon oxidation and nitrification [46]. Process characteristics, summarized by Diamadopoulos et al. [46] and Dollerer and Wilderer [81], resulted in a wide application for landfill leachate treatment [43,61,63,68,92]. Many authors have reported COD removals up to 75% (Table 8). Also, 99% NH_4^+-N removal has been observed by Lo [18] during the aerobic treatment of domestic leachates in a SBR with a 20–40 days residence time. The greater process flexibility of SBR is particularly important when considering landfill leachate treatment, which have a high degree of variability in quality and quantity [26].

3.1.2.1.2. Attached-growth biomass systems. Due to main problems of sludge bulking or inadequate separability [81] in conventional aerobic systems, a number of innovative aerobic processes, called attached-growth biomass systems, using biofilm, have been recently developed. These systems present the advantage of not suffer from loss of active biomass. Also nitrification is less affected by low temperatures [62] than in suspended-growth systems, and by inhibition due to high nitrogen content.

Trickling filters. This method has been investigated for the biological nitrogen lowering from municipal landfill leachate. Biofilters remain an interesting and attractive option for nitrification due to low-cost filter media [90]. Typical efficiencies of biofilters, encountered in literature, are presented in Table 8. In a recent work, above 90% nitrification of leachate was achieved in laboratory and on-site pilot aerobic crushed brick filters with loading rates between 100 and 130 mg NH_4^+-N L^{-1} day^{-1} at 25 °C and 50 mg NH_4^+-N L^{-1} day^{-1} even at temperatures as low as 5–10 °C, respectively [90]. In the last decade, maximum ammonia rejection of 97 and 75% in a trickling filter were,

Table 7
Lagooning performance

Feeding				Operational conditions				Performance removal (%)	Reference
COD (mg L^{-1})	BOD/COD	pH	From	Kind of lagoon	Size	T (°C)	HRT (days)		
5518	0.7	5.8	Landfill	Aerated lagoon	1000 m^3	–	>10	97 COD	[65]
–	–	–	Landfill	(1) Anaerobic pond (2) aerated lagoon (3) constructed wetlands (4) free water surface	(1) 400 m^3 (2) 4000 m^3 (3) 400 m^2 (4) 2000 m^2	–	40	60–95 COD	[66]
1182	0.26	–	Landfill	(1) Primary lagoon (2) aerated wetlands (3) final surge lagoon	(1) 113,400 m^3 (2) 4528 m^3	–	20	89 COD	[67]
765–3090	0.43–0.53	8.7–12.5	Landfill	(1) Aerated lagoon (2) polishing lagoon (laboratory-scale)	(1) 17 L (2) 9.7 L	19	(1) 16–22 (2) 9.1–12.6	55–64 COD	[36]
5050	0.25	8.38	Landfill	Non-aerated lagoon	9960 m^2	22.8	32	40 COD	[31]

Table 8
Different aerobic reactors performance

Feeding				Operational conditions			Performance removal (%)	Reference
COD (mg L^{-1})	BOD/COD	pH	From	Volume of reactor (L)	T (°C)	HRT (days)		
Activated sludge reactor								
4000 (BOD)	–	7	Landfill	>4	20–25	35	51.3 TOC	[71]
5000	0.6	5.95	Landfill	20	5–10	10 (SRT)	>92 COD	[72]
1000–4000	–	–	Landfill	470	–	–	–	[73]
1537	–	8	Coke-plant	6,700	–	1.5	96 COD	[74]
2000–4600	0.41–0.59	12–13	Landfill	5.9	21	6.25	46–64 COD	[75]
3176	0.33	–	Landfill	65,000 m^2	25	–	59 COD	[29]
2900	0.66	6.8–7.4	Landfill	0.5	24	2.75	75 COD	[76]
5000–6000	–	–	Landfill	5	25	0.5–2	97 COD 87.5 N-NH_4^+	[77]
2560	–	8	Landfill	30	25	–	–	[78]
200–1200 (NH_4^+)	–	7.5	Landfill	–	20	3.4 h	–	[79]
3130	0.56	–	Landfill	–	–	3	69 COD	[60]
270–1000	–	–	UASB reactor pre-treatment	3.35	5–10	10	50 COD	[37]
24,400	–	7.3	Landfill	40	23	–	80–90 COD	[22]
7439	–	8.22	Landfill	2	–	1	78–98 COD	[28]
5400–20,000	–	–	Municipal solid waste	9	–	4.5	85–89 COD	[80]
Sequencing batch reactor								
5295	0.49	9.1	Landfill	10–20	25	0.5	62 DOC	[81]
2560	0.07	8.6	Landfill	–	–	20–40	48–69 COD >99 NH_4^+	[18]
2110	0.4–0.5	6.9	Anaerobic lagoon pre-treatment	32	20	3.2	91 COD	[68]
1183	–	8	Landfill	45	–	1	6.7 COD	[33]
15,000	–	7.5	Landfill	8	40–50	–	75 COD	[82]
9500	–	7	Landfill	18.8	20	1.25	74 COD	[83]
7000	–	7	Synthetic wastewater	18.8	25	1.25	75 COD	[84,85]
5750	–	8.6	Landfill	5	25	–	62 COD	[43]
Moving-bed biofilm reactor								
2000–3000	0.41–0.59	12–13	Landfill	1	21	1	75 COD	[75]
1740–4850	0.05–0.1	9	Landfill	1.5	20	–	60 COD	[86]
800–1300	0.1	8	Landfill	0.22–0.6	5–22	2–5	20–30 COD	[12]
108	0.06	8	Landfill	4.5	20	–	42–57 DOC	[87]
800–2000	–		Landfill	5,000	17	4	20 COD	[88]
5000	0.2	>7.5	Landfill	8	–	20–24	81 COD 85 NH_3	[70]
480	0.05	7.7	Landfill (preozonation)	2	–	–	60–80 TOC	[89]
Trickling filter								
850–1350	0.1–0.2	8.0–8.5	Landfill	16,500	1.7–19.7	0.6–4.5	87 BOD	[40]
2560	–	8	Landfill	141	25	–	–	[78]
230–510	0.04–0.08	6.5–7	Landfill	9.4	5–25	2.1–9.6	90 NH_4^+	[90]

Table 9
Different anaerobic reactors performance

Feeding				Operational conditions			Performance removal (%)	Reference
COD (mg L^{-1})	BOD/COD	pH	From	Volume of reactor (L)	T (°C)	HRT (days)		
Digester								
4000 (BOD)	–	7	Landfill	>4	20–25	86	96 BOD	[71]
				2	24	30	53 COD	
37,000–66,660	0.4–0.6	–	Landfill	6	35	1–20	92.5 COD	[100]
1000–4000	–	–	Landfill	155	–	–	–	[73]
1537	–	8	Coke-plant	3300	–	0.75	95.7 COD	[74]
2560	–	8	Landfill	30	25	–	–	[78]
5100–8300	0.43–0.50	7.6–9.3	Landfill	1.25	15.5–35	2–10	56–70 COD	[101]
200–1200 (NH_4^+)	–	7.5	Landfill	–	20	1.72 h	–	[79]
800–2000	–	–	Landfill	900	17	0.72	20 COD	[88]
Anaerobic sequencing batch reactor								
546–5770 (TOC)	0.53	7.3–7.8	Landfill	2	35	10–1.5	73.9 TOC	[102]
15,000	–	7.5	Landfill (stabilized leachate)	8	40–50	–	75 COD	[82]
5750	–	8.6	Landfill	5	25	–	62 COD	[43]
Up-flow anaerobic sludge blanket reactor								
6649–15,425	–	7.6–8.7	Landfill	–	–	2.4	88 COD	
10,000–64,000	–	61–7.8	Landfill	3.5	15–35	0.6–0.1	82 COD	[103]
3000–4300	0.65–0.67	6.8–7.4	Landfill	0.38	11–24	0.4–1.4	45–71 COD	[76]
1500–3200	0.61–0.71	6.5–7.0	Landfill	40	13–23	0.96–1.30	65–75 COD	[104]
30,000	–	–	Landfill	4.6	30	0.75	82 COD	[105]
3800–15,900	0.54–0.67	7.3–7.8	Landfill	2	35	10–1.5	83 COD	[23]
3210–9190	–	6.9–9.0	Landfill	6.2	35	0.5–1	77–91 COD	[26]
9264–12,050	–	7.2	Anaerobic digestion plant sludge + septage + leachate	13.5	35	1.5–10	58 COD	[63]
24,400	–	7.3	Landfill	20	36	–	80–90 COD	[22]
35,000–50,000	0.5–0.6	5.6–7.0	Landfill	–	–	–	–	[24]
5400–20,000	–	–	Municipal solid waste	2.5	37–42	1.25	96–98 COD	[80]
Anaerobic filter								
14,000	0.7	5.8	Young landfill	3	21–25	2–4	68–95 COD	[16]
3750	0.3	6.35–6.58	Old landfill			0.5–1	60–95 COD	
5000–6000	–	–	Landfill	4	35	–	87.5 NH_4^+	[77]
Hybrid bed filter								
2000–3000	0.41–0.59	12–13	Landfill	2.5	21	62	75 COD	[75]
1800	0.53	6.8–7.4	Landfill	0.56	11	1.4	56 COD	[76]
19,600–42,000	–	6.5–7.5	Landfill	22	30	2.5–5	81–97 COD	[106]
1250–4490 (TOC)	0.53	7.3–7.8	Landfill	3.35	35	5.1–0.9	65.3 TOC	[102]
Fluidized bed reactor								
108	0.06	8	Landfill	4.5	20	–	42–57 DOC	[87]
1100–3800	–	–	Landfill	7.9	35	–	82 COD	[107]

respectively, claimed by Knox and Jones [93] and Martienssen and Schops [78].

Moving-bed biofilm reactor (MBBR) (or suspended-carrier biofilm reactor (SCBR) or fluidized bed reactor). MBBR process is based on the use of suspended porous polymeric carriers, kept in continuous movement in the aeration tank, while the active biomass grows as a biofilm on the surfaces of them. Mains advantages of this method compared to conventional suspended-growth processes seems to be: higher biomass concentrations, no long sludge-settling periods, lower sensitivity to toxic compounds [70] and both organic and high ammonia removals in a single process [86]. For instance, Welander et al. [94] reported nearly 90% nitrogen removal while the COD was around 20%. In case of treating high strength ammonia leachate, no inhibition of nitrification is encountered [12]. Moreover, the use of granular activated carbon (GAC) as porous material offers an appropriate surface to adsorb organic matter and optimised conditions for enhanced biodegradation [86]. Thus, a steady-state equilibrium is established between adsorption and biodegradation [95]. Imai et al. [87,96,97] developed an efficient biological activated carbon fluidized bed process. Nearly, 70% refractory organics were removed by coupling biological treatment and adsorption process [87]. After optimising the reactor operating regime, Horan et al. [86] and Loukidou and Zouboulis [70] proved possible to reach 85–90% ammonia reduction and 60–81% COD reduction.

3.1.2.2. Anaerobic treatment. An anaerobic digestion treatment of leachates allows to end the process initiated in the tip, being thus particularly suitable for dealing with high strength organic effluents, such as leachate streams from young tips [98]. Contrary to aerobic processes, anaerobic digestion conserves energy and produces very few solids, but suffers from low reaction rates [99]. Moreover, it is possible to use the CH_4 produced to warm the digester, that usually works at 35 °C and, under favourable conditions, for external purposes.

3.1.2.2.1. Suspended-growth biomass processes.

Digester. Performances of conventional anaerobic suspended-growth digester are reported in Table 9. Typical values of 80–90% and nearly 55% COD removals were reached in anaerobic lab-scale tank at 35 °C and ambient temperature, respectively [71,100,101].

Sequencing batch reactor. Some studies, presented in Table 9, revealed good performances of anaerobic sequencing batch reactors. These systems are able to achieve solid capture and organic lowering in one vessel, eliminating the need for a clarifier. Recently, nutrient reduction from pre-treated leachate was carried out using a lab-scale SBR by Uygur and Kargi [43]. Sequential anaerobic–aerobic operations resulted in COD, NH_4^+-N and PO_4^{3-}-P removal of 62%, 31% and 19%, respectively, at the end of cycle time (21 h). Also, in the initial period of the landfill, sufficient organic abatement in the anaerobic reactor through methanogenesis and denitrification, can enhance better nitrification in the following aerobic reactor. Therefore, anaerobic–aerobic system is recommended to bring down simultaneously organic and nitrogen matter [78,79,94]. For instance, Kettunen and Rintala [75] showed that COD removal was 35% in the anaerobic stage while in the combined process the COD and BOD_7 removals were up to 75% and 99%.

In last decades, the performance improvement of the existing anaerobic process was believed to be a promising option and so, high rate reactors have been designed in order to reduce long digestion time [69]. Except the conventional anaerobic suspended-growth reactor, UASB reactors are the main processes encountered in the literature (Table 9).

Up-flow anaerobic sludge blanket (UASB) reactor. UASB process is a modern anaerobic treatment that can have high treatment efficiency and a short hydraulic retention time [69]. UASB reactors, when they are submitted to high volumetric organic loading rate values [103], have exhibited higher performances compared to other kinds of anaerobic reactors. The process temperatures reported have generally been 20–35 °C for anaerobic treatment with UASB reactors. In these conditions, the average performance of COD decrease efficiency (Table 9) was always higher than 70% at ambient temperature (20–23 °C) and 80% at 35 °C. Up to 92% COD decreases were obtained by Kennedy and Lentz [26] at low and intermediate organic loading rates (between 6 and 19.7 g COD L^{-1} day^{-1}). Only a few studies have been conducted at temperatures between 11 and 23 °C [76,103,104,108] although leachates may be cooler than that, especially in cold countries. Kettunen and Rintala [104] showed that leachate can be treated on-site UASB reactor at low temperature. A pilot-scale reactor was used to study municipal landfill leachate treatment (COD 1.5–3.2 g L^{-1}) at 13–23 °C. COD (65–75%) and BOD_7 (up to 95%) removals were achieved at organic loading rates of 2–4 kg COD m^{-3} day^{-1}. Garcia et al. [103] concluded that COD rejection efficiency was not affected by temperature between 15 and 35 °C. These promising results show that high-rate treatment at low temperature may minimise the need for heating the leachate prior to treatment, which may thus provide an interesting cost-effective option [76]. The main disadvantages of such a treatment stay sensitivity to toxic substances [101].

3.1.2.2.2. Attached-growth biomass processes. Typical performances of such systems are presented in Table 9.

Anaerobic filter. The anaerobic filter is a high rate system that gathers the advantages of other anaerobic systems and that minimizes the disadvantages. In an up-flow anaerobic filter, biomass is retained as biofilms on support material, such as plastic rings [106]. For instance, Henry et al. [16] demonstrated that anaerobic filter could reduce the COD by 90%, at loading rates varying from 1.26 to 1.45 kg COD m^{-3} day^{-1}, and this for different ages of landfill. Total biogas production ranged between 400 and 500 L gas kg^{-1} COD destroyed and methane content between 75 and 85%.

Hybrid bed filter. It consists on an up-flow sludge blanket at the bottom and an anaerobic filter on top. This device acts as a gas–solid separator and enhances solid's retention without causing channelling or short-circuiting [102]. Enhanced performances of such a process results from maximization of the biomass concentration in the reactor. Nedwell and Reynolds [106] reported steady-state COD removal efficiencies of 81–97% under methanogenic digestion, depending upon organic loading rate. One drawback of hybrid reactor, as well as anaerobic filter, is the added cost of the support media.

Fluidized bed reactor. Suidan et al. [107] and Imai et al. [87,96,97] reported studies on carbon-assisted fluidized beds. The combined biodegradation and adsorption process provide a means for removing a variety of organic compounds [107]. Imai et al. [96] found that the biological activated carbon fluidized bed process was much more effective for treating old landfill leachate than the conventional one such as activated sludge and fixed film processes. The anaerobic treatability of this process is given in Table 9.

3.1.3. Physical/chemical treatment

Physical and chemical processes include reduction of suspended solids, colloidal particles, floating material, color, and toxic compounds by either flotation, coagulation/flocculation, adsorption, chemical oxidation and air stripping. Physical/chemical treatments for the landfill leachate are used in addition at the treatment line (pre-treatment or last purification) or to treat a specific pollutant (stripping for ammonia).

3.1.3.1. Flotation. For many years, flotation has been extensively used and focused on the decrease of colloids, ions, macromolecules, microorganisms and fibers [109]. However, until to date, very few studies have been devoted to the application of flotation for the treatment of landfill leachate. Recently, Zouboulis et al. [110] investigated the use of flotation in column, as a post-treatment step for removing residual humic acids (non-biodegradable compounds) from simulated landfill leachates. Under optimised conditions, almost 60% humic acids removal has been reached.

3.1.3.2. Coagulation–flocculation. Coagulation and flocculation may be used successfully in treating stabilized and old landfill leachates [7,111,112]. It is widely used as a pre-treatment [20,113,114], prior to biological or reverse osmosis step, or as a final polishing treatment step in order to remove non-biodegradable organic matter. Aluminum sulfate, ferrous sulfate, ferric chloride and ferric chloro-sulfate were commonly used as coagulants [113,115]. The application of bioflocculant, in comparison with traditional inorganics coagulants has been investigated by Zouboulis et al. [116], for the lowering of humic acids. It revealed as a viable alternative since 20 mg L^{-1} bioflocculant dosage was sufficient in providing more than 85% humic acid removal.

Several studies have been reported on the examination of coagulation–flocculation for the treatment of landfill leachates, aiming at process optimisation, i.e., selection of the most appropriate coagulant [20], identification of optimum experimental conditions and assessment of pH effect [113,117]. Synthesis of recent works, presented in Table 10, clearly reveal that iron salts are more efficient than aluminum ones, resulting in sufficient chemical oxygen demand (COD) reductions (up to

Table 10
Treatment effectiveness of landfill leachate with the use of coagulation/flocculation

COD (mg L^{-1})	BOD/COD	pH	From	Coagulant	Concentration range	Removal (%)	Reference
–	10,850 BOD	6.55	Landfill	$Ca(OH)_2$	0.3–0.6 g L^{-1}	8.2–23.5 COD	[118]
–	1500 BOD	–	Landfill	$Ca(OH)_2$	6 kg m^{-3}	57 COD	[119]
4000–8810	0.15	8.3	Landfill	$Ca(OH)_2 + Fe_2(SO_4)_3$	0.5–4.0 + 0–0.2 g L^{-1}	39 COD	[120]
4100	0.05	8.2	Landfill	$FeCl_3$ or $Al_2(SO_4)_3$	0.01–0.07 M	40–50 COD	[113]
6000–8200	0.11–0.17	–	Landfill	$Ca(OH)_2 + Al_2(SO_4)_3$	1.5 + 1.0 kg m^{-3}	42 COD	[121]
330 (biologically treated)	0.02	7.5	Landfill	$FeCl_3 + Al_2(SO_4)_3$	0.1–1.0 g L^{-1}	53 COD	[94]
282–417 TOC	–	–	Aerated lagoon	$FeCl_3$	0.8–1.0 g L^{-1}	38–48 TOC	[122]
782–1585	0.07–0.15	7.6–8.2	Landfill	$Al_2(SO_4)_3 + FeCl_3$	0.150 g L^{-1} Al + 0.05 g L^{-1}	20–35 COD	[10]
15,700	0.27	7.7	Landfill	$Fe_2(SO_4)_3$	0.3 g L^{-1} Fe	70 COD	[19]
–	–	–	Landfill	$Al_2(SO_4)_3 + FeCl_3$	0.738 + 1.136 g L^{-1}	55–70 color	[114]
1200–1500	0.04	6.8–7.5	Landfill	$FeCl_3$	0.2–1.2 g L^{-1}	39 COD	[123]
5350	0.2	7.9	Landfill	$FeCl_3 + Al_2(SO_4)_3$	1.0–5.0 g L^{-1}	75 COD	[20]
5000	<0.01	–	Landfill	$Fe_2(SO_4)_3 + Al_2O_3$	1.8–3.0 + 1.0–2.0 g L^{-1}	67 COD	[112]
7400–8800	0.05–0.06	8.5–9.0	Landfill	$FeCl_3$	0.01 M	40–90 COD	[44]
3460	0.04	8.2–8.5	Landfill	$Al_2(SO_4)_3$	0.7 g L^{-1}	10–25 COD	[7]
750	0.08	7.5	Landfill	Bioflocculant (*Rhizomonas*)	0.01–0.02 g L^{-1}	85 humic acid 40 COD	[116]

Table 11
Treatment effectiveness of landfill leachate with the use of chemical precipitation

COD ($mg\,L^{-1}$)	BOD/COD	pH	From	Precipitant	Removal (%)	Reference
1585 (young leachate)	0.07	8.2	Landfill	$Ca(OH)_2$ ($1\,g\,L^{-1}$)	27 COD	[11]
7511	0.19	8.22	Landfill	$MgCl_2{\cdot}6(H_2O) + Na_2HPO_4{\cdot}12(H_2O)$ ($Mg{:}NH_4{:}PO_4 = 1{:}1{:}1$)	40 COD 98 N-NH_4^+	[124]
65–1047	–	7.79–8.52	Landfill	$MgCl_2{\cdot}6(H_2O) + Na_2HPO_4{\cdot}12(H_2O)$ ($Mg{:}NH_4{:}PO_4 = 1{:}1{:}1$)	98 N-NH_4^+	[28]
35,000–50,000	0.5–0.6	5.6–7.0	Landfill	Struvite ($Mg{:}NH_4{:}PO_4 = 1{:}1{:}1$)	50 COD	[24]

50%), whereas the corresponding values in case of aluminum or lime addition were moderate (between 10 and 40%). Nevertheless, combination of coagulants [120] or addition of flocculants together with coagulants may enhance the floc-settling rate [113] and so the process performance (COD abatement up to 50%).

However, this treatment presents some disadvantages: consistent sludge volume is produced and an increase on the concentration of aluminum or iron, in the liquid phase, may be observed [7].

3.1.3.3. Chemical precipitation. In the case of leachate treatment, chemical precipitation is widely used as pre-treatment in order to remove high strength of ammonium nitrogen (NH_4^+-N). In a study, Li et al. [124] confirmed that the performance of a conventional activated sludge process could be significantly affected by a high concentration of NH_4^+-N. The COD removal declined from 95 to 79%, when the NH_4^+-N concentration in wastewater increased from 50 to 800 $mg\,L^{-1}$. So, many works have been initiated to investigate the feasibility of selectively precipitating NH_4^+-N (Table 11). Li et al. [28,124] precipitated ammonium ions as magnesium ammonium phosphate (MAP) with the addition of $MgCl_2{\cdot}6H_2O$ and $Na_2HPO_4{\cdot}12H_2O$ with a $Mg/NH_4/PO_4$ ratio of 1/1/1 at a pH of 8.5–9. Ammonium concentration was reduced from 5600 to 110 $mg\,L^{-1}$ within 15 min by this method. Yangin et al. [125] and Altinbas et al. [126] studied MAP precipitation after anaerobic pre-treatment of domestic wastewater and landfill leachate mixture. Maximum ammonia lowering was obtained as 66% at a pH of 9.3 at the stochiometric ratio whereas ammonia lowering reached to 86% at the same pH above the stochiometric ratio. In MAP precipitation at the stochiometric ratio and above the stochiometric ratio, ammonia concentration, in the up-flow anaerobic sludge blanket (UASB) reactor, was reduced to 31 $mg\,L^{-1}$ and 13 $mg\,L^{-1}$, respectively. Recently, struvite precipitation ($Mg{:}NH_4{:}PO_4 = 1{:}1{:}1$) was applied to anaerobically pre-treated effluents for ammonia removal [24]. Ammonium nitrogen depletion were observed as 85, 72 and 20% at pH of 9.2, 12 and 10–11, respectively.

Table 12
Treatment effectiveness of landfill leachate with the use of adsorption

COD ($mg\,L^{-1}$)	BOD/COD	pH	From	Adsorbent	Removal (%)	Reference
879–940 (biologically pre-treated)	0.03	7.5	Landfill	Granular activated carbon (columns)	91 COD	[128]
640	–	–	Landfill	Granular activated carbon (columns)	–	[127]
108	0.06	8	Landfill	Powdered activated carbon	–	[87]
800–2000	0.04–0.07	–	Landfill	Activated carbon (concentration range 2–10 $g\,L^{-1}$)	96 TOC	[88]
–	–	–	Landfill	Powdered activated carbon ($2\,g\,L^{-1}$)	55–70 color	[114]
625	0.3	7.9	Landfill	Peat	69 COD	[129]
9500	–	7	Landfill	Powdered activated carbon (0–2 $g\,L^{-1}$)	38 COD	[35]
1533–2580	0.03–0.04	7.5–9.4	Landfill	$CaCO_3$ (particle size range 2–4 mm)	90 COD	[41]
10,750–18,420	0.55	7.7–8.2	Landfill leachate + municipal sewage	Powdered activated carbon (concentration range 0.1–3.5 $g\,L^{-1}$)	–	[25]
7000	–	7	Synthetic wastewater	Powdered activated carbon (0–2 $g\,L^{-1}$)	90 COD	[84,85]
716–1765	–	7.58–7.60	Pilot plant	Granular activated carbon and resins	85 non-biodegradable COD (GAC) 59 non-biodegradable COD (resin)	[52]

3.1.3.4. Adsorption. The adsorption of pollutants onto Activated Carbon in columns [87,127,128] or in powder form [35,85,88,114] provides better reduction in COD levels than the chemicals methods, whatever the initial organic matter concentration (Table 12). The main drawback is the need for frequent regeneration of columns or an equivalently high consumption of powdered activated carbon (PAC). Adsorption by activated carbon has been used along with biological treatment for effective treatment of landfill leachate [25,47,128,130]. Non-biodegradable organics, inert COD and the color may be reduced to acceptable levels for biologically treated landfill leachate. Rodriguez et al. [52] studied PAC and different resins efficiency in the reduction of non-biodegradable organic matter from landfill leachate. Activated carbon presented the highest adsorption capacities with 85% COD decrease and a residual COD of 200 mg L^{-1}.

Recently, simultaneous adsorption and biological treatment has been tested. For instance, pre-treated leachate (coagulation–flocculation and air stripping of ammonia) was subjected to biological treatment in an aeration tank operated in repeated fed-batch mode in the presence of adsorbent (PAC and powdered zeolite) [84]. Nearly 87% and 77% COD removals were achieved with PAC and zeolite concentrations of 2 g L^{-1}, respectively. Other adsorbent media have been studied. Heavey [129] used a pre-treated peat as the treatment medium. Almost 100% removal of both BOD and ammonia, and 69% removal of COD were achieved. Moreover, treatment rates of 36 g BOD m^{-2} day^{-1} and 11 g ammonia m^{-2} day^{-1}, similar with those obtained by high cost aerobic lagoons systems, were noticed. In 1988, McLellan and Rock [131] already concluded that filtration through peat can be used only as a pre-treatment process to reduce metal concentrations prior to a conventional treatment. Finally, limestone has been proven effective in removing metals from wastewaters. Aziz et al. [41] indicated that 90% of Fe could be removed from semi-aerobic landfill leachate by limestone filter, based on retention time of 57.8 min and surface loading of 12.2 m^3 m^{-2} day^{-1}.

3.1.3.5. Chemical oxidation. Chemical oxidation is a widely studied method for the treatment of effluents containing refractory compounds such as landfill leachate. Growing interest has been recently focused on advanced oxidation processes (AOP). Most of them, except simple ozonation (O_3), use a combination of strong oxidants, e.g. O_3 and H_2O_2, irradiation, e.g. ultraviolet (UV), ultrasound (US) or electron beam (EB), and catalysts, e.g. transition metal ions or photocatalyst. Table 13 lists typical AOP systems currently reported in the literature. All these processes have been recently reviewed by Wang et al. [13]. Authors confirmed that AOP, adapted to old or well-stabilized leachate, are applied to:

- oxidize organics substances to their highest stable oxidation states being carbon dioxide and water (i.e., to reach complete mineralization),
- improve the biodegradability of recalcitrant organic pollutants up to a value compatible with subsequent economical biological treatment.

Table 13
List of typical AOP systems [132]

Homogeneous system
With irradiation
O_3/ultraviolet (UV)
H_2O_2/UV
Electron beam
Ultrasound (US)
H_2O_2/US
UV/US
H_2O_2/Fe^{2+}/UV (photo-Fenton's)
Without irradiation
O_3/H_2O_2
O_3/OH^-
H_2O_2/Fe^{2+} (Fenton's)
Heterogeneous systems
With irradiation
TiO_2/O_2/UV
TiO_2/H_2O_2/UV
Without irradiation
Electro-Fenton

Performance of each process can be evaluated thanks to key parameters (COD, BOD, BOD/COD, oxidant dose...) summarized in Tables 14 and 15. Although many of the previous researchers using ozonation have demonstrated the effectiveness in eliminating COD (reduction is about 50–70% in most cases) [89,133,138,149] most of them only used this process as tertiary treatment prior to discharge in the environment. Sometimes the treatment efficiency on stabilized leachates has been moderate [7]. After 1 h of ozonation (1.3–1.5 g O_3/g COD degraded), only 30% COD depletion was observed by Rivas et al. [44]. COD lowering can be greatly enhanced combining oxidants (H_2O_2/O_3) (Table 12) or adding an irradiation system (H_2O_2/UV) (Table 13). Wable et al. [143], Bigot et al. [133] and Schulte et al. [144] reported organic matter removal efficiency as high as 90% for the O_3/H_2O_2 process. Concerning the H_2O_2/UV process, the BOD_5/COD ratio has been increased significantly from 0.1 to 0.45 by Qureshi et al. [142]. Also, Steensen [138] reported 85–90% of COD reduction with a biologically pre-treated leachate. Fenton and photo-Fenton processes allow COD decrease efficiency of, respectively, 45–75% and 70–78%. In term of biodegradability improvement, BOD/COD ratios close to 0.5 after oxidation have been reported in recent works using Fenton process [21,151]. Finally, a few papers reported photocatalytic treatment [27,42,153,154] or electron-beam radiation treatment [60] of organic components from landfill leachates even at laboratory-scale. These technologies have been applied to treat or degrade principally humic substances.

However, common drawbacks of AOP is the high demand of electrical energy for devices such as ozonizers, UV lamps, ultrasounds, which results in rather high treatment costs [21]. Also, for complete degradation (mineralization) of the pollutants to occur, high oxidant doses would be required, rendering the process economically expensive. Silva et al. [7] applied high ozone doses (until 3.0 g L^{-1}) to attain significant toxicity decrease. Furthermore, some intermediate oxidation prod-

Table 14
O_3, O_3/H_2O_2 and O_3/UV treatments of leachates (updated from Wang et al. [13])

COD ($mg L^{-1}$)	BOD ($mg L^{-1}$)	pH	COD removal (%)	BOD/COD after treatment	O_3/COD (g/g)	H_2O_2/O_3 (g/g)	UV (W)	Reference
Ozonation								
1610	–	–	44	–	1.3			[132]
2300	210	8	62	–	1.5			[133]
2300	210	3	50	–	0.5			
2300	210	8	50	–	1			
740	240	–	–	–	–			[134]
4000	230	8.5	25	–	0.53			[135]
640	205 DOC	–	–	0.4	1.28–1.92			[127]
460	–	–	71	–	1.8			[136]
1050	–	8.5	67	–	1.7			
500	30	7	–	140 $mg L^{-1}$ BOD_5	0.11			[137]
300–1200	<10	7.0–8.0	80	–	3			[138]
151	5	8.1	33	0.35	–			[87]
330	<8	7.5	35	0.15	3.5			[94]
1585	111	8.2	23–32	–	1.7			[11]
518	–	8.3	66	–	1.7			[139]
895	43	8.2	30	0.11	1.11			[140]
3500	25	8.2	67	0.21	0.7			[141]
480	25	7.7	>50	0.25	0.5			[89]
14,600	2920	7.8	56	0.32	3.1			[142]
2300–4970	290–850	7.90–9.02	30	0.25	1.3–1.5			[117]
6500	500	8.1	15	0.5	1.2 $g L^{-1}$ (O_3 dose)			[34]
3460	150	8.3	2.5–48	–	0.1–3 $g L^{-1}$ (O_3 dose)			[7]
O_3/H_2O_2								
2000	–	–	95	–	3.5	0.4		[143]
600	–	–	92	–	3.3	0.4		
2000	160	8.4	92	0.13	1.5	0.3		[133]
–	–	8	97	–	2.5 $g L^{-1}$ (O_3 dose)	1		[144]
–	–	8	70	–	–	0.5		
895	43	8.2	28	0.14	0.76	–		[145]
1360	<5	8.4	93	0.32	1.5	0.3		
480	25	7.7	40	0.13	0.05–0.5	0.25–1		[89]
O_3/UV								
1280	100	2	54	–	–		100	[146]
1280	100	2	47	–	–		500	
2300	210	8	50	–	1		15	[133]
430 TOC	–	–	51 TOC	–	0.1 $g L^{-1}$ (O_3 dose)		300	[147]
26,000	2920	7.8	63	0.32	3.5		1500	[142]
26,000	2920	7.8	61	0.35	4.7		1500	

ucts can actually raise the toxicity of the leachate. Among these processes and according to Lopez et al. [21], Fenton's process seems to be the best compromise because the process is technologically simple, there is no mass transfer limitation (homogeneous nature) and both iron and hydrogen peroxide are cheap and non-toxic. But Fenton's process required low pH and a modification of this parameter is necessary.

3.1.3.6. Air stripping. Nowadays, the most common method for eliminating a high concentration of NH_4^+-N involved in wastewater treatment technologies is air stripping. High levels of ammonium nitrogen are usually found in landfill leachates, and stripping can be successful for eliminating this pollutant, which can increase wastewater toxicity [5]. If this method is to be efficient, high pH values must be used and the contaminated gas phase must be treated with either H_2SO_4 or HCl. Performances of this process can be evaluated in term of ammonia-nitrogen removal efficiency (Table 16). Marttinen et al. [5] reported a 89% ammonia reduction at pH = 11 and 20 °C within 24 h retention time. High rates of ammonia removal have been achieved by Cheung et al. [155] in spite of high initial ammonia concentration (0.5–0.7 $g N L^{-1}$). Their results showed that 93% of 309–368 $mg L^{-1}$ ammonia-nitrogen were removed in free stripping tanks with 1 day retention time. In recent works, 85 and 99.5% of ammonia reduction has been, respectively, attained by Ozturk et al. [24] and Silva et al. [7]. But a major concern about ammonia air stripping is the release of NH_3 into the atmosphere so as to cause severe air pollution if ammonia cannot be properly absorbed with either H_2SO_4 or HCl. Others drawbacks are the calcium carbonate scaling of the stripping tower, when lime is used for pH adjustment, and the problem of foaming which imposes to use a large stripping tower [124].

Table 15
H_2O_2/UV, H_2O_2/Fe^{2+} and H_2O_2/Fe^{2+}/UV in leachates treatment (updated from Wang et al. [13])

COD ($mg\,L^{-1}$)	BOD ($mg\,L^{-1}$)	pH	COD removal (%)	BOD/COD after treatment	UV (W)	H_2O_2 ($g\,L^{-1}$)	Fe^{2+} ($mg\,L^{-1}$)	Reference
H_2O_2/UV								
760	–	–	22	–	150	3.4		[144]
760	–	3	99	–	150	3.4		
1000–1200	<10	3.0–4.0	90	–	15	0.5		[138]
1000–1000	<10	3.0–4.0	85	–	150	0.5		
1280	100	2	57	–	100	–		[146]
1280	100	2	59	–	500	–		
430 TOC	–	–	42 TOC	–	300	–		[147]
26,000	2920	3	79	0.37	1500	5.19		[142]
26,000	2920	3	91	0.4	1500	13		
26,000	2920	3	96	0.45	1500	26		
H_2O_2/Fe^{2+}								
–	–	3	50	–		1.6	–	[144]
1050–2020	50–270	4	60	–		0.2	600–800	[148]
1200	–	–	63	0.15		–	–	[77]
1150	3–5	3	70	–		2.44	56	[149]
2000	87	3.5	69	0.58		1.5	120	[150]
330	<8	7.5	72	0.3		10 $mL\,L^{-1}$	20	[94]
282–417 TOC	–	3	49–76 TOC	–		1	1250	[122]
–	–	3	55	–		2.2	–	
1500	30	3.5	75	–		1.65	645	[111]
Old leachate	–	–	–	–		1	1000	[114]
1800	225	3	52	0.22		1.5	2000	[151]
1800	225	4.5	45	0.27		1.2	1500	
1500	75	6	70	–		0.2	300	
1500	75	8.5	14	–		0.2	300	[17]
10,540	2300	8.2	60	0.5		1	830	[21]
H_2O_2/Fe^{2+}/UV								
1150	3–5	3	70		500–1000	1.15	56	[149]
1150	–	3.2	70		UVA	1.15	72	[152]
440	–	2.7	78		UVA	0.44	30	

3.1.4. Conclusion on conventional treatments

During many years, conventional biological treatments and classical physico-chemical methods are being considered as the most appropriate technologies for manipulation and management of high strength effluents like landfill leachates. When, treating young leachate, biological techniques can yield a reasonable treatment performance with respect to COD, NH_3-N and heavy metals. When treating stabilized (less biodegradable) leachate, physico-chemical treatments have been found to be suitable as a refining step for biologically treated leachate, in order to remove organic refractory substances. The integrated chemical–physical–biological processes (whatever the order) ameliorates the drawbacks of individual processes contributing to a higher efficacy of the overall treatment.

However, with the continuous hardening of the discharge standards in most countries and the ageing of landfill sites with more and more stabilized leachates, conventional treatments (biological or physico-chemical) are not sufficient anymore to reach the level of purification needed to fully reduce the negative impact of landfill leachates on the environment. It implies that new treatment alternatives species must be proposed. Therefore, in the last 20 years, more effective treatments based on membrane technology has emerged as a viable treatment alternative to comply and pending water quality regulations in most countries.

Table 16
Treatment effectiveness of landfill leachate with the use of air stripping

From	NH_4^+-N ($mg\,L^{-1}$)	Time of stripping (h)	NH_4^+-N removal (%)	Reference
Landfill	556–705	24	76–93	[155]
Landfill	74–220	24	89	[5]
Synthetic wastewater	1270	0.75	45	[35,83]
Landfill	1025	17	85	[24]
Landfill	800	120	99.5	[7]

3.2. New treatments: the use of membrane processes

Microfiltration, ultrafiltration, nanofiltration and reverse osmosis are the main membrane processes applied in landfill leachates treatment.

3.2.1. Microfiltration (MF)

MF remains interesting each time that an effective method is required to eliminate colloids and the suspended matter like, for instance, in pre-treatment for another membrane process (UF, NF or RO) or in partnership with chemical treatments. But, it cannot be used alone. Only Piatkiewicz et al. [156], in a polish study, reported the use of MF as prefiltration stage. No significant retention rate (COD reduction between 25 and 35%) was achieved (Table 17).

3.2.2. Ultrafiltration (UF)

UF is effective to eliminate the macromolecules and the particles, but it is strongly dependant on the type of material constituting the membrane. UF may be used as a tool to fractionate organic matter and so to evaluate the preponderant molecular mass of organic pollutants in a given leachate. Also, tests with membrane permeates may give information about recalcitrance and toxicity of the permeated fractions. Except Tabet et al. [39], UF was eliminated as a primary means for treating landfill leachate due to drastic existing regulations. These authors used membranes close to nanofiltration, leachate had a low organic matter content and local water standards were not so strict. However, Syzdek and Ahlert [157] suggested that UF might prove to be effective as a pre-treatment process for reverse osmosis (RO). UF can be used to remove the larger molecular weight components of leachate that tend to foul reverse osmosis membranes. Table 18 summarizes studies including an UF step. The elimination of polluting substances is never complete (COD between 10 and 75%). More recently, UF has been applied to biological post-treatment of landfill leachate [33]. Several hybrid processes such as activated sludge–ultrafiltration–chemical oxidation and activated sludge–ultrafiltration–reverse osmosis have been tested. Same authors demonstrated that 50% of the organic matter could be separated by the UF step alone.

Finally, UF membranes have been successfully used in full scale membrane bioreactor plants [30]. High treatment levels for landfill leachate have been achieved in such a process.

3.2.2.1. Membrane bioreactors. The combination of membrane separation technology and bioreactors has led to a new focus on wastewater treatment. It contributes to very compact systems working with a high biomass concentration and achieving a low sludge production with an excellent effluent quality. Membrane bioreactors have been widely applied at full scale on industrial wastewater treatment and some plants have been adapted to leachate treatment [30]. However, few research studies are related to landfill leachate purification by membrane bioreactors (Table 19). Pirbazari et al. [6] used a hybrid technology known as the ultrafiltration-biologically active carbon (UF-BAC) process that amalgamates adsorption, biodegrada-

Table 17
Treatment effectiveness of landfill leachate with the use of microfiltration

Operating conditions						Feeding			Performance		Reference
Material/geometry	Cut-off	Surface (m^2)	T (°C)	Velocity ($m\,s^{-1}$)	P (bar)	From	COD ($mg\,L^{-1}$)	pH	Flux	COD removal (%)	
Polypropylene/tubular (Membrana GmbH/Accurel)	0.2 μm	0.11	20	4.1–4.3	–	Landfill	2300	7.5	–	25–35 (retention rate)	[156]

Table 18
Treatment effectiveness of landfill leachate with the use of ultrafiltration

Operating conditions						Feeding			Performance		Reference
Material/geometry	Cut-off	Surface (m^2)	T (°C)	Velocity ($m\,s^{-1}$)	P (bar)	From	COD ($mg\,L^{-1}$)	pH	Flux ($L\,h^{-1}\,m^{-2}$)	COD removal (%)	
Substituted olefin, aromatic, polymer, polyelectrolyte complex, cellulose acetate (Amicon)	0.5–300 kDa	–	–	–	–	Landfill	14,000–17,000 (TOC)	7.0	30–180	–	[157]
Cellulosic/tubular (Memtek Corp.)	dp = 0.2 μm	0.0065	20–45	–	20–22 p.s.i.	Landfill	8300–9500	7.0	–	95–98	[6]
PVC/flat	20–55 kDa	0.0155	25	2.5	3	Landfill	1660	8.6	–	50	[33]
Polysulfone/tubular	300 kDa	0.025									
Polysulfone/tubular (Membrana GmbH/UltraPES)	50–80 kDa	0.15	20	4.1–4.3	–	Landfill	1700	–	–	5–10	[156]

Table 19
Membrane bioreactor effectiveness for the treatment of landfill leachates

Feeding				Operational conditions				Performance	Reference
COD ($g\,L^{-1}$)	BOD/COD	pH	From	Kind of reactor	Volume of reactor (m^3)	T (°C)	HRT (days)	COD removal (%)	
4000	0.2	–	Landfill	Industrial scale	180	–	–	>90	[158]
2750–3105	0.48	6.5–7.5	Landfill	Stirred tank/biologically active carbon process	15	28–30	3–4	95–98 TOC	[6]
2740–3200	0.51								
–	–	–	Landfill	Pilot research	–	–	–	90	[30]

Table 20
Treatment effectiveness of landfill leachate with the use of nanofiltration

Operating conditions						Feeding			Performance		Reference
Material/geometry	Cut-off	Surface (m^2)	T (°C)	Velocity ($m\,s^{-1}$)	P (bar)	From	COD ($mg\,L^{-1}$)	pH	Flux ($L\,h^{-1}\,m^{-2}$)	COD removal (%)	
Spiral wound (Desal)	50% NaCl 1 ppm	–	–	–	8.5	Landfill	–	–	7–12	97.5–99	[73]
Organic/tubular (PCI Membrane Systems)	–	0.04	25	2.8	15–30	Landfill	142 TOC	–	55–75	55–60 TOC	[160]
Polyacrilonitrile/flat (Koch–Weizmann)	450 Da	0.007	25	1–5	0–15	Landfill	550–2295	7.4–7.8	18	60	[161,162]
Polysulfone/flat (Koch–Weizmann)	450 Da	0.007							52	75	
Oxide de zirconium/tubular (Koch–Weizmann)	1000 Da	0.125							57	65	
Polyacrilonitrile/tubular (Koch–Weizmann)	450 Da	0.049	25	3	20	Landfill	500	7.5	80	74	[38]
Polysulfone/tubular (Koch–Weizmann)	450 Da								60	80	
Polymer/flat sheet (Desal)	200–300 Da	0.0045	25	3	6–8	Landfill	200–600	7.3–7.9	–	52–66	[5]

Table 21
Treatment effectiveness of landfill leachate with the use of reverse osmosis

Operating conditions				Feeding			Performance		Reference
Material/geometry	Surface (m^2)	T (°C)	P (bar)	From	COD ($mg\,L^{-1}$)	pH	Flux ($L\,h^{-1}\,m^{-2}$)	Removal (%)	
Composite/tubular (PCI Membrane Systems)	0.013	20	40	Landfill	335–925	–	3–48	>98 COD	[163]
Tubular/spiral wound	–	25	40	Landfill (biological pre-treatment)	1301	–	30	99 COD	[29]
Spiral wound	–	28	20–53	Landfill	0–1.749	6	–	96–98 COD	[32]
Cellulose acetate/flat (Osmonics)	0.0155	25	27.6	Landfill	846	8.8	–	93 COD	[33]
Spiral wound	–	20	–	Landfill	1820	5.6–6.6	–	–	[156]
Polyamide/spiral wound (Filmtec)	6.7	–	–	Landfill (MBR pre-treatment)	211–856	–	–	97 COD	[45]
Polyamide (Desal)	0.0044	–	60	Landfill (evaporation pre-treatment)	200.5	8	20.7–29	86–90 COD	[3]
Polyamide/DT-module (Pall)	7.6	15.5–31.8	9–70.5	Landfill	–	4.8–7.0	47.2–102.8 L/h/module	50–85 COD	[168]
	7.6	–	3–11			5.0–5.9	50–105.8 L/h/module	80–90 COD	
	7.9	–	26–174			–	–	–	
Spiral wound	2	30	25	Landfill	1700	8	32	99 COD	[24]
			55		3000		58	89 COD	

tion and membrane filtration. The process efficiencies were in the range of 95–98% in terms of TOC reduction, and exceeded 97% for specific organic pollutants. Contrary to conventional systems, organisms such as nitrifiers or organisms which are able to degrade slowly biodegradable substances are not washed out of the system and no loss of process activity occurs.

3.2.3. *Nanofiltration (NF)*

NF technology offers a versatile approach to meet multiple water quality objectives, such as control of organic, inorganic, and microbial contaminants. NF studied membranes are usually made of polymeric films with a molecular cut-off between 200 and 2000 Da. The high rejection rate for sulfate ions and for dissolved organic matter (Table 20) together with very low rejection for chloride and sodium reduces the volume of concentrate [159]. Few studies mention the use of NF to treat landfill leachates [5,38,73,162–164]. Nearly 60–70% COD and 50% ammonia were removed by NF, whatever membrane material and geometry (flat, tubular, or spiral wounded), with an average velocity of $3\ m\ s^{-1}$ and a transmembrane pressure between 6 and 30 bar. Physical methods were used in combination with nanofiltration and it was found satisfactory for removal of refractory COD from the leachate used. COD removal was 70–80% [162].

However, successful application of membrane technology requires efficient control of membrane fouling. A wide spectrum of constituents may contribute to membrane fouling in leachates nanofiltration: dissolved organic and inorganic substances, colloidal and suspended particles [162]. In particular, natural organic matter fouling has recently gained interest [165,166].

3.2.4. *Reverse osmosis (RO)*

RO seems to be one of the most promising and efficient methods among the new processes for landfill leachate treatment. In the past, several studies, performed both at lab and industrial scale, have already demonstrated RO performances on the separation of pollutants from landfill leachate [163,167]. Values of the rejection coefficient referred to COD parameter and heavy metal concentrations higher than 98 and 99%, respectively, were reported (Table 21). Tubular and spiral wounded modules were the first medium used in the early RO systems for the purification of landfill leachate starting in 1984. An innovative technology was introduced to this market in 1988 with great success: the disc-tube-module (DT-module) developed by Pall-Exekia. Thanks to open channel module, systems can be cleaned with high efficiency with regard to scaling, fouling and especially biofouling [169]. In 1998, Peters [170] reported that more than 80% of the total capacity installed for leachate purification by RO use a DT-module (Germany, the Netherlands, Switzerland, North America. . .).

Depending on the salt content of the feed water and the operation time between the cleaning cycles, the operating pressure ranges between 30 and 60 bar at ambient temperature and the specific permeate flux reach $15\ L\ h^{-1}\ m^{-2}$ [171]. The average specific energy demand is low with less than $5\ kW\ h\ m^{-3}$ of permeate for a recovery rate of 80% [169].

Table 22
Effectiveness of treatment vs. leachate characteristics

Process	Character of leachate			Average removal (%)					Residues
	Young	Medium	Old	BOD	COD	TKN	SS	Turbidity	
Transfer									
Combined treatment with domestic sewage	Good	Fair	Poor	Depending on domestic water treatment plant					Excess biomass
Recycling	Good	Fair	Poor	>90	60–80	–	–	–	–
Lagooning	Good	Fair	Poor	80	40–95	>80	30–40	30–40	Sludge
Physico/chemical									
Coagulation/flocculation	Poor	Fair	Fair	–	40–60	<30	>80	>80	Sludge
Chemical precipitation	Poor	Fair	Poor	–	<30	<30	30–40	>80	Sludge
Adsorption	Poor	Fair	Good	>80	70–90	–	–	50–70	–
Oxidation	Poor	Fair	Fair	–	30–90	–	–	>80	Residual O_3
Stripping	Poor	Fair	Fair	–	<30	>80	–	30–40	Air–NH_3 mixture
Biological									
Aerobic processes	Good	Fair	Poor	>80	60–90	>80	60–80	–	Excess biomass
Anaerobic processes	Good	Fair	Poor	>80	60–80	>80	60–80	–	Excess biomass
Membrane bioreactor	Good	Fair	Fair	>80	>85	>80	>99	40–60	Excess biomass
Membrane filtration									
Ultrafiltration	Poor–Fair			–	50	60–80	>99	>99	Concentrate
Nanofiltration	Good	Good	Good	80	60–80	60–80	>99	>99	Concentrate
Reverse osmosis	Good	Good	Good	>90	>90	>90	>99	>99	Concentrate

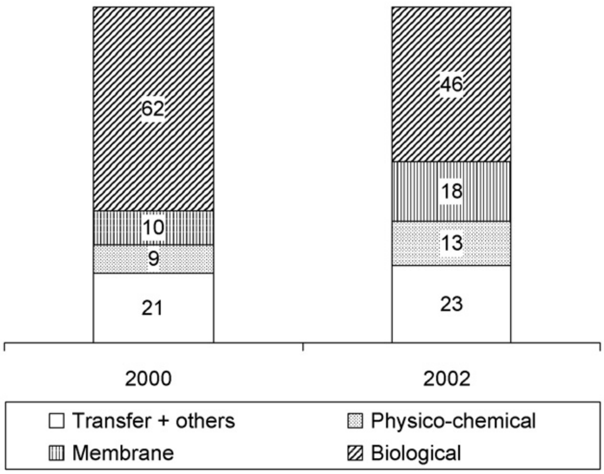


Fig. 4. Landfill leachate treatment distribution, in France [2].

However, two issues have been identified, and remain today, as major drawbacks for the implementation of pressure-driven membrane processes, and particularly RO, to landfill leachate treatment: membrane fouling (which requires extensive pre-treatment or chemical cleaning of the membranes, results in a short lifetime of the membranes and decreases process productivity) and the generation of large volume of concentrate (which is unusable and has to be discharged or further treated). In the early 1990s, steady improvement of membrane technology and striving for high water recoveries in landfill leachate treatment resulted in development of a high pressure RO system based on the DT-module and operating at transmembrane pressures of 120 and 200 bar. An adapted process permits to reduce certain salt fractions by controlled precipitation. This means an increase of the permeate recovery from about 80% to 90% with a concentration factor of 10 and a reduction of concentrate volume [172].

4. Discussion and conclusion

Optimal leachate treatment, in order to fully reduce the negative impact on the environment, is today's challenge. But, the complexity of the leachate composition makes it very difficult to formulate general recommendations. Variations in leachates, in particular their variation both over time and from site to site, means that the most appropriate treatment should be simple, universal and adaptable. The various methods presented in the previous sections offer each advantages and disadvantages with respect to certain facets of the problem.

Suitable treatment strategy depends on major criteria:

- *The initial leachate quality*. Table 22 summarizes the effectiveness of treatment process according to key leachate characteristics: COD, BOD/COD and age of the fill. The knowledge of these specific parameters may help to select suitable treatment processes for the lowering of organic matter present in leachate.
- *The final requirements given by local discharge water standards*. Year after year, the recognition of landfill leachate impact on environment have forced authorities to fix more and more stringent requirements for pollution control. Even by combining biological and physico/chemical processes, only partial destruction of contaminants will be achieved. Due to the so-called "hard COD", new regulations will not be reached. In recent years, membrane filtration has emerged as a viable treatment alternative to comply with existing and pending water quality regulations.

Today, the hardening of landfill regulations, controls and management hamper an efficient conventional treatment (such as aerobic or anaerobic biological methods, physico/chemical treatments), which appears under-dimensioned or does not allow to reach the specifications required by the legislator. So that, membrane processes, and most particularly RO offers the best solution, and have been proved to be the more efficient, adaptable and indispensable means of both:

- achieving full purification (rejection rates of 98–99% for RO),
- solving the growing problem of water pollution.

In Fig. 4 concerning leachate treatments distribution, French case clearly reflects the worldwide trend, namely a marked increase of pressure-driven membrane processes in comparison with biological treatment plants.

However, landfill leachate RO feasibility is highly conditioned by the control of concentrate treatment costs and the choice of the feed pre-treatment mode in order to reduce membrane fouling. Residue production, which constitute a capital environmental concern, still remain major hurdle, since it is usually unusable and has to be discharged, further treated or landfilled. The transport to an incineration plant equipped for the burning of liquid hazardous waste remains the preferred option (in spite of many controversies) but leads to high treatment costs. Others possibilities are slowly gaining importance [170,173,174]:

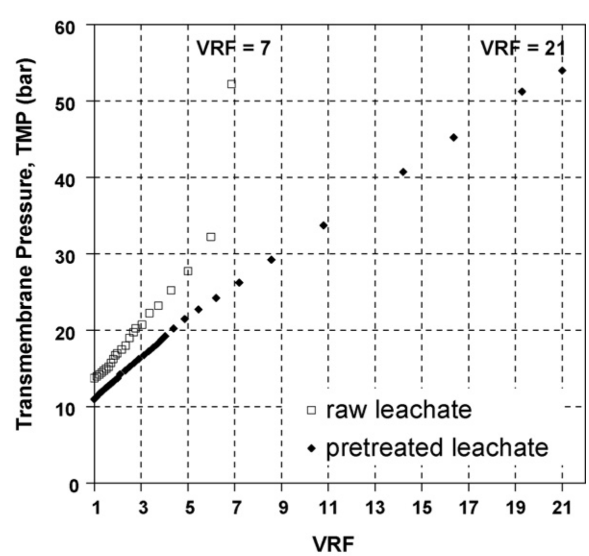


Fig. 5. RO treatment, in a concentration mode and constant permeate flux ($10\,L\,h^{-1}\,m^{-2}$), of raw and pre-treated leachate ("lime + RDVPF")—spiral wound membrane (Koch Membrane Systems), 20 °C.

- the solidification of residues with different materials, like fly ash or sludges from wastewater treatment plants, and disposal on the landfill itself,
- controlled reinjection of the concentrate into changing areas of the landfill.

Methods to reduce the cost of treatment residues must be developed or improved with respect to ecological and economical requirements, biogas capture must be promoted, because it permits interesting exploitation cost reductions.

Moreover, techniques to prevent or control membrane fouling need to be further investigated (suitable pre-treatment choice, modifications affecting surface membrane roughness or hydrophilicity/hydrophobicity, cleaning of membrane surface. . .). Biological pre-treatment are often proved ineffective as RO pre-treatment [15,45,175].

On the contrary, lime precipitation appears like a promising option for the pre-treatment of RO membranes and the removal of colloidal particles and organic macromolecules that are the principal RO foulants of landfill leachates [15,176,177]. In the same way, microfiltration and ultrafiltration have proved to be suitable, provided that they are preceded by physico/chemical process as lime precipitation [157,178,179].

Although lime precipitation is traditionally used to eliminate the temporary hardness of the water by decarbonation, it has been shown by a number of studies – focusing mainly on underground or surface water treatment – to be able of removing by co-precipitation certain high molecular weight organic molecules such as humic and fulvic acids, responsible for irreversible membrane fouling [180–183]. Pre-treatment by lime precipitation therefore appears as a promising approach for the leachate treatment by RO. However, whereas in the above-mentioned studies the separation of the precipitate was done through decantation, here the solid/liquid separation upstream of the RO unit is performed using a rotatory drum vacuum precoat filter (RDVPF). This type of filter has already proved efficient for the separation of inorganic solid phases during the treatment of nuclear effluents, and during clarification of grape must. The use of such a filter for the clarification of lime pre-treated leachates would present several advantages over decantation:

- the guarantee of a constant quality of the pre-treated leachate, thanks to the use of a filter medium in place of the decanter,
- the elimination by the filtering layer of the non-settleable small-sized particles,
- the reduction of the volumes of sludge generated,
- a reduction of the size of the facility by suppressing the decanter.

Preliminary experiments showed that the addition of lime at optimum doses of 5 g L^{-1} triggers a mechanism of decarbonation of the leachates, that is, a 15–40% decrease in the salinity through elimination of the temporary hardness linked to the presence of calcium and magnesium and through massive precipitation of $CaCO_3$. This pre-treatment also makes it possible to remove 20–30% of the COD, essentially refractory organic macromolecules (PM > 50,000 g mol^{-1}) such as humic acids, according to a mechanism of co-precipitation—mechanism validated by Scanning Electron Microscopy visualization.

The RDVPF is particularly efficient in separating of the precipitated phase – essentially composed of calcium carbonate – generated by the lime pre-treatment. The continuous de-scaling of the filtration surface by means of the micrometric advancing knife allows relatively high fluxes – ranging from 650 to 1000 L h^{-1} m^{-2} – to be reached with this type of facility. The total sludge production at the end of the RDVPF step reaches on average 5 g dry sludge L of pre-treated leachate, resulting on average at 85% from the formation of $CaCO_3$, at 10% from the co-precipitation of organic macromolecules and at 5% from the

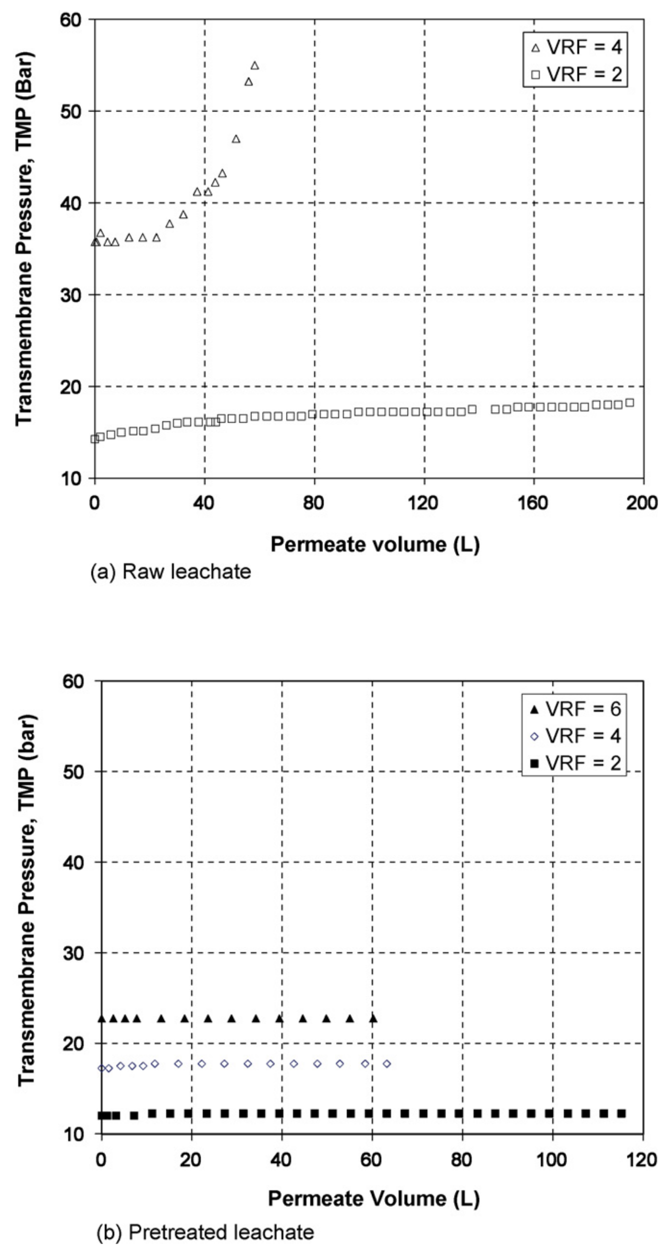
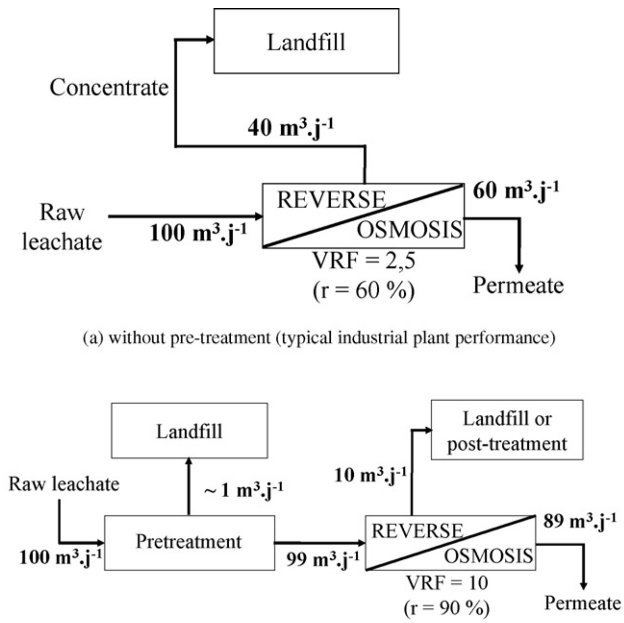


Fig. 6. RO treatment, in a constant volumetric reduction factor (VRF) mode and constant permeate flux (10 L h^{-1} m^{-2}), of (a) raw leachate and (b) pre-treated leachate (“lime + RDVPF”)—spiral wound membrane (Koch Membrane Systems), 20 °C.



(a) without pre-treatment (typical industrial plant performance)

(b) with pretreatment "lime + RDVPF" (expectations)

Fig. 7. Comparison of RO plant performance with (a) raw (industrial operation) and (b) pre-treated (expected improvement with "lime precipitation + filtration on RDVPF" pre-treatment) landfill leachate.

scraping of the diatomaceous layer. The interesting characteristics of the sludges obtained (siccity, dehydratability, stability, low volume and very good pelletability) make it possible to consider an easy and well-advised storage of these sludges at the municipal solid waste landfill (MSWLF) site.

In comparison with ultrafiltration, the operation and capital costs of such a pre-treatment "lime + RDVPF" are, respectively, reduced for 80 and 50%. Volumes of residues are also largely reduced. Moreover, applying this pre-treatment makes it possible to considerably reduce the operating costs of the RO unit by reducing both the working pressures (by 8–20%) and the concentrate volumes generated by operating at up to 3 times higher VRF (Fig. 5). As shown in Fig. 6, this pre-treatment also eliminates almost all the fouling, probably due to the humic acids co-precipitated during the lime precipitation. A significant decrease in the frequency of membrane washings and in the use of cleaning chemicals can be expected. Considering a stabilized leachate with an average conductivity of 15 mS cm^{-1}, the process combination would make it possible to reach global conversion rates close to 90%, rather than the current 60% conversion rates at most industrial sites (Fig. 7). As for the fate and the handling of the low volume of concentrate generated by RO, several solutions can be considered: (i) storing it at the site, which would entail a premature increase in the salt load of the tip and (ii) eliminating it by incineration, at a cost of 8–10€/m^{-3} (Soumont, France) This combination of processes has been subject to a European patent pending process [184].

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