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Activated sludge process *versus* rotating biological contactors in WWTPs: Evaluating the influence of operation and sludge bacterial content on their odor impact

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ABSTRACT

Two municipal wastewater treatment plants (WWTPs), based on activated sludge process (ASP) and rotating biological contactors (RBC) as biological treatments, were comparatively evaluated in terms of their operational conditions, bacterial content and physicochemical characteristics of their derived sludge (SL) to determine their influence on odor impact. The average values of influent wastewater flow, inlet chemical oxygen demand (COD) and COD removal efficiency were (ASP-WWTP *vs.* RBC-WWTP): 447 *vs.* 689 m^3/d, 300 *vs.* 423 mg/L and 88.28 *vs.* 83.17%, respectively. Regarding the global odor emissions, ASP-WWTP and RBC-WWTP had a similar odor emission rate (11,177 ou_E/s and 12,784 ou_E/s, respectively), with sludge thickening and dewatering being the major sources of odor in both facilities. Proteobacteria, Bacteroidetes and Firmicutes were the three predominant phyla in both WWTPs, representing the 83% in ASP-SL and the 97% in RBC-SL. RBC-SL showed lower bacterial biodiversity than ASP-SL. The higher odor concentration from the sludge handling activities in RBC-WWTP were linked to the significative increments in the abundance of Porphyromonadaceae, Clostridiales, Lachnospiraceae (obligate anaerobe) and Moraxellaceae (aerobic) families compared to ASP-WWTP. However, when odor emissions were evaluated per equivalent inhabitant (EI), a higher value was obtained for ASP-WWTP (16.22 ou_E/s·EI) compared to RBC-WWTP (6.84 ou_E/s·EI).

1. Introduction

The generation of wastewater is an inevitable consequence of human and industrial activities due to urbanization, industrialization and population growth. In contemporary society, protecting the environment, reducing energy consumption, preserving raw materials and minimizing waste generation are becoming increasingly important. For this reason, the implementation and development of wastewater treatment technologies are required to enhance their efficiency, so that they not only comply with necessary standards and a low energy footprint but also boost economic feasibility and environmental sustainability (Sikosana et al., 2019). Wastewater treatment is a key operation based on physical, chemical and biological processes that allow several pollution parameters to be reduced (Kalbar et al., 2012). Whether to comply with environmental regulations or to avoid negative impacts on nearby bodies of water, it is convenient to know the basics of the wastewater treatment (*i.e.*, multiple factors including technical, economic, regulatory,

Abbreviations: ASP, activated sludge process; BOD_5, biochemical oxygen demand at 5 days; C/N, carbon/nitrogen ratio expressed as VS/N-TKN; COD, chemical oxygen demand; DO, dissolved oxygen; EI, equivalent inhabitant; F/M, food-to-microorganism ratio; FS, fixed solids; HRT, hydraulic retention time; IC, soluble inorganic carbon; MLTSS, mixed liquor total suspended solids; MLVSS, mixed liquor volatile suspended solids; N-NH_4^+, ammoniacal nitrogen; N-TKN, total Kjeldahl nitrogen; N-TN_s, soluble total nitrogen; OC, odor concentration; OD_{20}, cumulative oxygen demand at 20 h; OER, odor emission rate; ou_E, European odor units; P-P_2O_5, phosphorus content expressed as P_2O_5; Q_e, effluent waterflow; Q_i, influent waterflow; RBC, rotating biological contactors; SBR, sequencing batch reactor; SL, sludge; SOER, specific odor emission rate; $SOUR_{max}$, maximum specific oxygen uptake rate; SRT, solids retention time; TC, soluble total carbon; TOC, soluble total organic carbon; TSS, total suspended solids; VOCs, volatile organic compounds; VS, volatile solids; VSCs, volatile sulfur compounds; WWTP, wastewater treatment plant

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microbiological and operational behavior as well as limitations) and the advantages and disadvantages of existing technologies to achieve the required treatment goals in accordance with the size and the needs of the population served (Prasse et al., 2015; Ullah et al., 2020).

The design of a wastewater treatment plant (WWTP) mainly depends on the pollutant load to be treated in terms of biochemical oxygen demand (BOD_5) or chemical oxygen demand (COD) (Dominguez and Gujer, 2006). One of the main concerns in the treatment of wastewater is selecting the most appropriate technology, which depends on various factors such as pollutant load, energy efficiency, governmental and regulatory factors, and environmental impact, including the emission of volatile organic compounds (VOCs) and nuisance odors (Ullah et al., 2020). In this context, a technology's appropriateness is not limited to its physical properties (referred to as 'hard aspects'), but also includes knowledge about transfer mechanisms, microbiological aspects and waste generation, among other factors.

Among the various operations of a WWTP, biological treatment is considered the most complex and influential process in the elimination of soluble pollutants in the integral wastewater treatment (Liu et al., 2019). It has the purpose of digesting and transforming biodegradable organic material into cells or cellular tissue and other harmless by-products, such as carbon dioxide and water or even mineral salts (*e.g.*, nitrates and sulfates) (Chahal et al., 2016; Saeed et al., 2021). Compared to chemical treatments, biological treatments have proved to be more efficient in resource consumption, having lower chemical and energy requirements, which can be translated into economic savings and environmentally friendly methods for municipal wastewater treatment (Oh et al., 2010; Wei et al., 2018).

More specifically, activated sludge process (ASP) and rotating biological contactors (RBC) are conventional biological treatments commonly implemented in medium-sized municipal WWTPs (Sikosana et al., 2019; Wee Seow et al., 2016; Zhang et al., 2020). The main microorganisms involved in these treatments are protozoa, fungi, algae, filamentous organisms and bacteria, the latter making up 90–95% of the process (Aonofriesei and Petrosanu, 2007). For ASP, biological oxidation takes place in an aerated biological reactor or aeration tank, where the biological culture (mixed liquor) is in contact with the wastewater with the purpose of transforming soluble organic matter through oxidation reactions (Waqas et al., 2020). The above-mentioned biological culture is formed by a large number of microorganisms grouped into flocs, the most abundant being the bacterial population, in the sequence Proteobacteria, Actinobacteria, Bacteroidetes and Firmicutes (Yu and Zhang, 2012). On the other hand, for RBC, the microorganisms are attached onto a rigid surface (disks) to form a biofilm, which is responsible for the degradation of organic pollutants (Rana et al., 2018). In the latter case, bacteria constitute about 98% of the total community (Ziembińska-Buczyńska et al., 2019). Moreover, a high biomass concentration (about 200 g dry weight/m^2) is maintained on the disc surface, having a metabolic effect similar to that produced by a concentration of 4000–6000 mg MLTSS/L in ASP (MLTSS stands for mixed liquor total suspended solids) (Jácome et al., 2015). Regarding the two technologies' technical aspects, it is important to highlight that ASP has a higher removal efficiency of organic pollutant load than processes based on a biofilm culture on a solid support. In turn, the energy consumption of RBC is about 60% lower than in ASP (*i.e.*, lower electricity consumption to remove the same amount of organic matter). However, the maintenance costs of systems equipped with RBC are approximately 35% higher than with ASP (Ferrer et al., 2008).

Given that biological treatment is very efficient in removing biodegradable soluble organic pollutants (except for emerging pollutants such as antibiotics, drugs, endocrine disruptors, industrial additives, *etc.*), and much of the removed matter is transformed into biological sludge, large amounts of sludge are generated (García et al., 2021). Sewage sludge is an inevitable by-product produced in the wastewater treatment process, in which numerous nutrients and organic materials are retained (*i.e.*, 24–34% of total NPK); recycling them as fertilizer can promote sustainability and a circular economy (Kominko et al., 2019). Sludge production in WWTPs varies widely, from 31 to 85 g of dry solids per equivalent inhabitant (EI) and per day (g TS/EI·d), related to the amount of settleable solids in raw wastewater and whose total suspended solids (TSS) content is in a range of 50–60 g TSS/EI·d or 110–170 g TSS/m^3 of treated wastewater (Tchobanoglous et al., 2003). In ASP, organic matter is oxidized by heterotrophic microorganisms, whereby the biodegradable particulate fraction is subjected to hydrolysis to generate new cellular biomass (cryptic growth), while the endogenous fraction (8–20%) remains and accumulates in the sludge. This endogenous fraction has many cellular capsules that sediment poorly, increasing the settling volume and the odor-producing compounds caused by sludge storage, such as volatile organic sulfur compounds, VOCs, volatile fatty acids, H_2S, NH_3, *etc.* (Bhatla, 1975; Carrera-Chapela et al., 2014; Karageorgos et al., 2010). In the case of RBC, the low concentration of sloughed-off biofilm improves sludge settlement quickly and freely, minimizing the odoriferous emission surface, *i.e.*, with a lower sludge volume and a shorter hydraulic retention time (HRT), allowing the efficient separation and management of the sludge (Cortez et al., 2008; Hassard et al., 2015; Jácome et al., 2015; Waqas and Bilad, 2019). In this respect, odor emissions are inherently associated with wastewater management, being volatile sulfur compounds (VSCs) primarily responsible for unpleasant odors. Certainly, previous studies have reported that VSCs can account for up to 80–90% of malodorous compounds in WWTPs (Li et al., 2021; Omri et al., 2011). Unpleasant odors (VSCs and VOCs) are closely associated with the generation, treatment and management of solid waste (such as biological sludge) as well as with the different types of treatment to which wastewater is subjected. However, most of the literature available deals with the odor emissions derived from biological treatment technologies individually, such as sequencing batch reactors (SBR) (Li et al., 2021), ASP (Dinçer et al., 2020; Kim et al., 2014; Varela-Bruce and Antileo, 2021; Zwain et al., 2020) or RBC (Cortez et al., 2008), rather than evaluating them comparatively. The presence of odor-causing compounds reduces the quality of ambient air and can result in the discomfort of the WWTP's workers as well as the residents of neighboring areas. It is well known that long-term exposure to odorants can cause psychological stress and symptoms such as headaches, nausea, insomnia, respiratory affections and even cancer (Byliński et al., 2019).

Therefore, considering the importance of identifying and controlling the odor emissions related to wastewater management, two municipal WWTPs with different and widespread biological treatments (ASP and RBC) were jointly evaluated in this study with the aim of: (i) comparing the physicochemical and respirometric characteristics of the sewage sludge derived from both types of plants; (ii) taxonomically identifying the bacteria responsible for the degradation processes to compare the most predominant families in the sludge generated in both types of plants and connecting them to their operational conditions; and finally (iii) performing a comparative evaluation of the odor emissions derived from both types of WWTPs, analysing the odor contributions of the different treatment stages. To the best of our knowledge, this integral approach to improve understanding of the characteristics of odor emissions depending on the biological treatment implemented has not been previously reported in the literature. This information may help wastewater managers to select the most appropriate wastewater treatment technology for minimizing the impact of odors when building new WWTPs close to population centers or to make decisions regarding odor abatement in existing ones, with the consequent social and environmental benefits.

Table 1
Wastewater treated flow and operational variables of both WWTPs.

	ASP-WWTP		RBC-WWTP	
	Influent (i)	Effluent (e)	Influent (i)	Effluent (e)
Q (m^3/d)	447 ± 331	394 ± 262	689 ± 186	677 ± 169
COD (mg/L)	300 ± 44	40 ± 9	423 ± 62	72 ± 15
BOD_5 (mg/L)	93 ± 22	5 ± 3	163 ± 39	12 ± 3
TSS (mg/L)	131 ± 32	6 ± 2	286 ± 67	19 ± 4

ASP-WWTP, wastewater treatment plant with activated sludge process; BOD5, biochemical oxygen demand at 5 days; COD, chemical oxygen demand; Q, waterflow; RBC-WWTP, wastewater treatment plant with rotating biological contactors; TSS, total suspended solids.

2. Material and methods

2.1. Location, characteristics and treatment capacity of the WWTPs

In this study, which was carried out in 2019, two municipal WWTPs were selected according to the most predominant biological technologies implemented in the province of Cordoba (Spain): ASP and RBC. The sampled facilities (satellite images are shown in Fig. S1, Supplementary material) were the following:

- **ASP-WWTP**: This facility operates with ASP as biological treatment (based specifically on extended aeration) and is located in Espiel (Cordoba). It has a design treatment capacity of 2424 EI. The average urban wastewater flow treated in 2019 was approximately 447 m^3/d (Table 1).
- **RBC-WWTP**: This facility has a biological treatment based on RBC and a design capacity of 6000 EI. It is situated in Villaviciosa de Cordoba (Cordoba) and treated around 689 m^3/d of urban wastewater in 2019 (Table 1).

As can be observed in Scheme 1, both WWTPs have the same unit operations, except for the type of biological treatment conducted. Moreover, ASP-WWTP has an anoxic reactor to perform wastewater denitrification, with internal and external recirculation between this reactor and the extended aeration tank. The HRT for the biological treatment of ASP-WWTP is approximately 20 h (about 14 h for the aerobic reactor and 6 h for the anoxic reactor), while the value of this parameter is considerably lower in the case of RBC-WWTP (≈ 4 h). More operational conditions such as energy consumption, dissolved oxygen (DO) and sludge concentration in the reactor, food-to-microorganism ratio (F/M), and solids retention time (SRT) can be observed in Table S1 (Supplementary material). Sludge thickening and mechanical sludge dewatering by centrifugation are common to both WWTPs.

Table 1 shows the mean values of the operational variables considered in the sampled WWTPs in 2019, which are COD (mg/L), BOD_5 (mg/L) and TSS (mg/L). These are variables of relevant interest established by Directive 91/271/CEE (EU Council, 1991) in non-sensitive areas (such is the case of Espiel and Villaviciosa de Cordoba) and were determined in accordance with the Standard Methods of the APHA (2017). This information as well as the above-mentioned annual mean values and operational conditions were provided by EMPROACSA, which is the wastewater manager in the province of Cordoba.

The treatment capacity of both plants in 2019 was also calculated in term of EI. To do so, Directive 91/271/CEE (EU Council, 1991) establishes that 1 EI has a biodegradable organic load with a BOD_5 equivalent to 60 g O_2/day. Therefore, the number of EIs depends on the influent waterflow to be treated (Q_i) and the inlet's average BOD_5, calculated according to Eq. (1):

$$EI = \frac{Q_i(L/d) \cdot BOD_5(gO_2/L)}{60\ gO_2/d} \tag{1}$$

where EI is the number of equivalent inhabitants, Q_i is the average influent waterflow and BOD_5 is the average biochemical oxygen demand at the facilities' inlets.

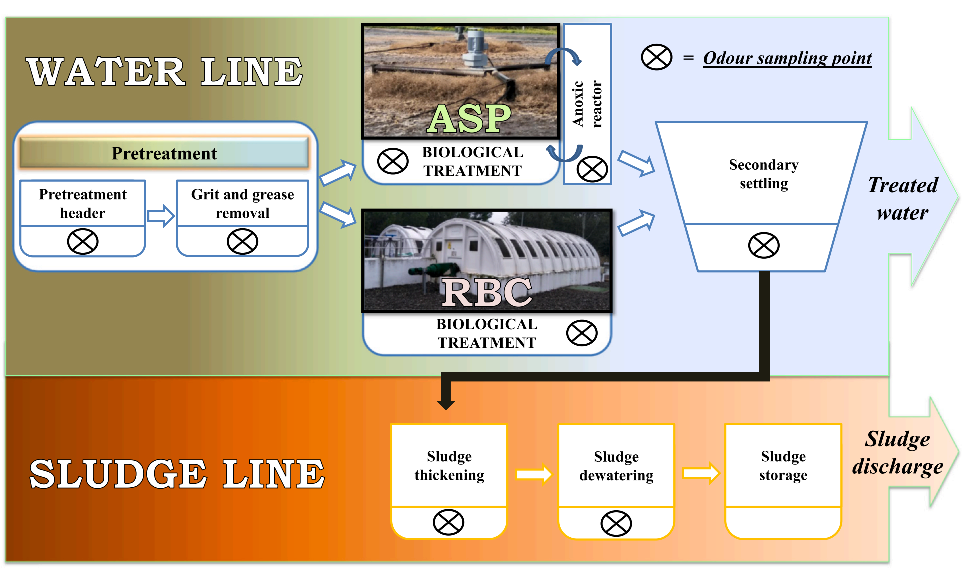
In accordance with the information provided about the two WWTPs (Table 1), the received pollutant load in both WWTPs, expressed in terms of chemical oxygen demand (kg COD/d), biochemical oxygen demand (kg BOD_5/d) or total suspended solids (kg TSS/d), was estimated through Eq. (2):

$$\text{Received pollutant load (kg/d)} = Q_i \cdot C_i \tag{2}$$

where Q_i is the average influent waterflow (L/d) and C_i refer to the average inlet concentrations (COD, BOD_5 or TSS, kg/L).

The COD, BOD_5 and TSS average removal efficiencies of these facilities were also calculated according to Eq. (3):

$$\text{Average removal efficiency}(\%) = \frac{Q_i{\cdot}C_i - Q_e{\cdot}C_e}{Q_i\,C_i} \tag{3}$$



Scheme 1. General flow diagram of the WWTPs evaluated (ASP-WWTP and RBC-WWTP).

where Q_i is the average influent waterflow, Q_e is the average effluent waterflow and C_i and C_e refer to the average inlet and outlet concentrations (COD, BOD_5 or TSS), respectively.

2.2. Physicochemical characterization and respirometric analysis of the sludge

The sludge evaluated for both technologies was subjected to physicochemical analysis following the methodology established by the United States Department of Agriculture and the United States Composting Council (2002). The sludge was identified as ASP-SL for ASP-WWTP and RBC-SL for RBC-WWTP. Fixed solids (FS, %), volatile solids (VS, %), total Kjeldahl nitrogen (N-TKN, %), ammoniacal nitrogen (N-NH_4^+, %) and phosphorus content (P-P_2O_5, %) were analysed in the solid fraction, while pH, conductivity (μS/cm), soluble total nitrogen (N-TN_s, %), soluble total carbon (TC, %), soluble inorganic carbon (IC, %), and soluble total organic carbon (TOC, %) were measured in the aqueous extract (1:25 v/v ratio). All variables were determined in triplicate, showing the mean values and standard deviation. It is important to highlight that the C/N ratio was also determined as VS/N-TKN.

In order to evaluate the microbial activity of the sludge of both facilities, the maximum specific oxygen uptake rate ($SOUR_{max}$, mg O_2/g VS·h) and cumulative oxygen demand at 20 h (OD_{20}, mg O_2/g VS) were determined under standardized conditions by using a static-liquid respirometer patented by the Department of Inorganic Chemistry and Chemical Engineering of the University of Cordoba, Spain (P2004–02908) and developed by Chica et al. (2003). The evolution of SOUR determines the reaction rate to stabilize organic matter. For its part, OD_{20} refers to the oxygen consumption per gram of VS added (measured on a dry basis) in the respirometric test, and it is a respirometric variable proportional to the biodegradability of the substrate (Martín et al., 2018). The respirometric analysis consisted of two 1-L reactors (Erlenmeyer flasks) with a discontinuous aeration system that maintained the oxygen concentration within a previously established interval (6.8—7.0 mg O_2/L). In such analysis, sludge samples were introduced into the reactors, both containing a nutritional medium with the same composition: 10 mL of $CaCl_2$ (27.50 g/L), 10 mL of $FeCl_3$ (0.25 g/L) and 10 mL of $MgSO_4$ (22.50 g/L). In addition, 30 mL of a phosphate buffer solution (K_2HPO_4 (8.50 g/L), $Na_2HPO_4.7\,H_2O$ (33.40 g/L), NH_4Cl (1.70 g/L)) were added to each reactor, providing a buffered medium at a pH of 7.2. In order that all oxygen uptake was attributed to the biological oxidation of the carbonaceous matter, thiourea was also added (1 g per reactor). This compound has a strong ability to inactivate ammonia-oxidizing bacteria, and therefore has a long history of being used as a nitrification inhibitor (Wang et al., 2017). Once all the substances were added, the reactors were made up to 1 L with distilled water. Because the metabolic activity of microorganisms depends on the temperature, both Erlenmeyer flasks were placed in a water thermostatic bath at 30 ºC. Finally, the reactors were also kept under constant agitation (300 rpm) to homogenize the solution and facilitate the transfer of oxygen (Chica et al., 2003). Each respirometric test was carried out by using 10 g of wet sludge. Moreover, each sample was analysed in duplicate, yielding the mean values and standard deviation.

2.3. Odor sampling and analysis

The odor sources sampled were the same in both WWTPs and can be observed in Scheme 1, comprising pretreatment header (roughing operations), grit and grease removal, biological treatment (ASP or RBC), secondary settling, sludge thickening and sludge dewatering. According to the guideline VDI 3880 (2011), all of them were passive odor sources. Sludge storage was not sampled in any of the WWTPs because it takes place in closed silos.

The CSD30 (manufactured by Olfasense GmbH) was the sampling device used for collecting odor samples in 10-L Nalophan® sampling bags. The ventilated sampling hood (Olfasense GmbH) was connected to the CSD30 device for sampling all passive odor sources mentioned above. Both sampling devices were fully compliant with the standard EN 13725 (2003) and the guideline VDI 3880 (2011). The main characteristics and operation of these devices can be found in Toledo et al. (2018a). The sampling time was 30 min per sample. All gaseous samples were analysed within 6 h in order to minimize the permeation and/or adsorption of odorants through/on the sampling bag walls, as proposed by the guideline VDI 3880 (2011).

Dynamic olfactometry (EN 13725, 2003) was the method used to quantify the odor concentration (OC). The units of this variable are ou_E/m^3 (European odor units per cubic meter). According to the European standard, 1 ou_E/m^3 is defined as the amount of odorant that, when evaporated into 1 m^3 of gas air at standard conditions, causes a physiological response from a panel (detection threshold) equivalent to that of n-butanol (reference gas) evaporated into 1 m^3 of neutral gas. A TO8 olfactometer (Olfasense GmbH), based on the 'Yes/No' method, was used to determine the OC of each sample. The group of panelists consisted of 4 people, each of whom was selected based on their sensitivity to the n-butanol reference gas as described in EN 13725 (2003). Each sample was analysed in duplicate and the OC was calculated as the geometric mean of the odor threshold values of each panelist, multiplied by the square root of the olfactometer dilution factor. All odor concentration data were expressed in accordance with the reference conditions described in EN 13725 (2003) (*i.e.*, 20 ºC, 101.3 kPa on a wet basis).

The OC results were necessary to subsequently determine the specific odor emission rate (SOER) and the odor emission rate (OER) of the sampled odor sources. The first variable was determined by Eq. (4):

$$\mathrm{SOER}\,(\mathrm{ou_E/m^2\,s}) = \frac{OC \cdot Q_{air}}{A_H} \tag{4}$$

where OC is the odor concentration (ou_E/m^3), Q_{air} is the airflow rate circulating through the sampling hood ($2.08 \cdot 10^{-3}$ m^3/s) and A_H is the covered area of the above-mentioned hood (1 m^2).

The SOER determination was the first step in estimating the OER from passive sources, which was calculated according to Eq. (5):

$$\mathrm{OER}\,(\mathrm{ou_E/s}) = SOER{\cdot}A \tag{5}$$

where the $SOER$ is the specific odor emission rate ($ou_E/m^2{\cdot}s$) and A is the odor emission surface of each odor source (m^2), which is detailed in Table 4.

Finally, a global odor emission rate (ou_E/s) was estimated for each of the WWTPs, calculated as the sum of the OER values of the different odor sources sampled.

2.4. DNA and genomic analysis

2.4.1. DNA isolation from WWTP sludge

The sludge of each WWTP was collected and kept at −20 °C. Samples were washed 3 times with Tris-HCl (pH 8.3) 50 mM; NaCl 200 mM; Na_2EDTA 5 mM; Triton X-100 0.05%. DNA was isolated from 700 mg of 3 independent batches using aluminum sulfate to avoid the interference of humic acids (Dong et al., 2006). Prior to the 16S analysis, DNA concentration was determined spectrophotometrically. DNA integrity was checked by running the samples on a 1% agarose gel electrophoresis. Furthermore, the removal of PCR inhibitors was checked with a PCR test using bacterial 16 S standard primers (forward: 5′-TGGTGGAATTTCCTGTGTAGCGGTGAA-3′ and reverse 5′-GCAACGCGAAGAACCTTACCTGGCCTT-3′). Amplifications were performed in a 50 µl reaction volume, using 0.5 µM (each) primer and approximately 20 ng of template DNA. The PCR cycling protocol consisted of the following: 95 °C for 1 min, 30 cycles 95 °C

Table 2
Treatment capacity, average removal efficiencies of organic matter according to the implemented biological treatment (ASP or RBC) and sludge production in each WWTP evaluated.

Treatment capacity		
	ASP-WWTP	**RBC-WWTP**
Capacity in year 2019 (EI)	689	1870
Average removal efficiency		
	ASP-WWTP	**RBC-WWTP**
COD (%)	88.28	83.17
BOD_5 (%)	95.58	92.74
TSS (%)	96.05	93.60
Sludge production (dry basis) provided by the wastewater manager		
	ASP-WWTP	**RBC-WWTP**
Sludge amount (kg/d)	102.85	134.81

ASP-WWTP, wastewater treatment plant with activated sludge process; BOD5, biochemical oxygen demand at 5 days; COD, chemical oxygen demand; EI, equivalent inhabitant; RBC-WWTP, wastewater treatment plant with rotating biological contactors; TSS, total suspended solids.

Table 3
Physico-chemical and respirometric characterization of sewage sludge from the WWTPs evaluated.

	ASP-SL	RBC-SL
pH	7.66 ± 0.08	7.12 ± 0.01
Conductivity (μS/cm)	122 ± 1	521 ± 4
FS (%)	38.31 ± 0.75	23.34 ± 1.38
VS (%)	61.69 ± 1.52	76.66 ± 1.49
N-TKN (%)	5.24 ± 0.03	8.29 ± 0.35
N-NH_4^+ (%)	0.79 ± 0.04	2.76 ± 0.09
N-TN_s (%)	0.43 ± 0.01	2.54 ± 0.03
C/N	11.78 ± 0.37	9.25 ± 0.58
P-P_2O_5 (%)	5.41 ± 1.72	4.93 ± 1.28
TC (%)	1.67 ± 0.01	6.13 ± 0.01
IC (%)	1.01 ± 0.01	1.69 ± 0.01
TOC (%)	0.66 ± 0.01	4.44 ± 0.01
$SOUR_{max}$ (mg O_2/g VS·h)	41 ± 2	17 ± 1
OD_{20} (mg O_2/g VS)	152 ± 3	160 ± 2

ASP-SL, sludge from the wastewater treatment plant with activated sludge process; C/N, carbon/nitrogen ratio expressed as VS/N-TKN; FS, fixed solids; IC, soluble inorganic carbon; N-NH_4^+, ammoniacal nitrogen; N-TKN, total Kjeldahl nitrogen; N-TNs, soluble total nitrogen; OD20, cumulative oxygen demand at 20 h; P-P2O5, phosphorus content expressed as P2O5; RBC-SL, sludge from the wastewater treatment plant with rotating biological contactors; SOURmax, maximum specific oxygen uptake rate; TC, soluble total carbon; TOC, soluble total organic carbon; VS, volatile solids.
Note: All variables, except pH and conductivity, are expressed on a dry basis.

for 15 s, 55 °C for 30 s, 72 °C for 30 s, and a final elongation at 72 °C for 7 min

2.4.2. Library construction, sequencing, gene prediction and taxonomy assignment

To identify the bacteria present in both sludges, DNA was independently amplified and sequenced using the specific kit for 16S sequencing (Ion Torrent System) at the Central Service for Research Support (SCAI) of the University of Cordoba (Spain). The results were analysed using Ion Reporter™ 5.0 software for Ion 16 S™ Metagenomics. The primers for the V3 region always provided the largest identification numbers and were thus used to estimate the bacterial community.

3. Results and discussion

3.1. Operating conditions of the two WWTPs: removal efficiencies of organic matter

The odor impact generated by a WWTP on surrounding areas is closely related to the organic load of the wastewater treated by the plant, sludge production and its subsequent treatment. In this sense, the operational variables of both WWTPs, provided by EMPROACSA, should be carefully considered. As can be observed in Table 1, the average influent waterflow in the case of RBC-WWTP was considerably higher than in ASP-WWTP (about 1.5-fold). This fact, together with the higher average concentration values of COD, BOD_5 and TSS of the influent wastewater in RBC-WWTP compared to ASP-WWTP (1.41, 1.75 and 2.18-fold, respectively), led to the first facility also receiving a higher pollutant load than ASP-WWTP, expressed in terms of chemical oxygen demand (291.46 *vs.* 133.99 kg COD/d), biochemical oxygen demand (112.31 *vs.* 41.54 kg BOD_5/d) or total suspended solids (197.07 *vs.* 58.51 kg TSS/d). This marked difference between the organic loads of the wastewater treated in the WWTPs also led to an important difference in the treatment capacity between both facilities (year 2019). Thus, Table 2 shows that RBC-WWTP had a higher treatment capacity (1870 EI) compared to ASP-WWTP (689 EI). Regarding the average removal efficiencies of organic matter, Table 2 also shows the COD, BOD_5 and TSS average removal efficiencies (%) of the WWTPs evaluated. In this respect, the results revealed a small difference between the two WWTPs, ASP technology being the most efficient. It should be noted that the biodegradation of organic matter involves the generation of sewage sludge (in solid stage), a liquid effluent that contains non-biodegradable organic matter and malodorous gaseous emissions with the potential to affect the health and well-being of nearby population areas (Lebrero et al., 2011). The proportion of each type of effluent depends on the technology applied to develop the biological treatment. In a comparative manner, ASP is an intensive technology, which requires an elevated energy supply, generating highly mineralized sewage sludge. As can be observed in Table 2, sludge production (on a dry basis) using ASP was 102.85 kg/d (year 2019). On the other hand, RBC is a technology in which microorganisms are expanded on biofilms, and organic matter is biodegraded in alternating anoxic and aerobic conditions. The sludge generated by using this treatment is less mineralized than that generated by ASP. In RBC-WWTP, sludge production in 2019 reached 134.81 kg/d (on a dry basis).

3.2. Assessment of physicochemical and respirometric results

As previously mentioned, the characteristics of sludge are closely related to the odor emissions generated during its treatment. Therefore, the physicochemical and respirometric characterization of the sewage sludge samples for the two WWTPs is shown in Table 3. Among the most relevant differences, the lower organic matter content of ASP-SL in comparison with RBC-SL, expressed as VS (%) or as TOC (%) in the soluble form, can be observed. The differences associated with nitrogen content (N-TKN, N-NH_4^+ or N-TN_s) in the two sludges were marked by the presence of an anoxic reactor to perform wastewater denitrification in ASP-WWTP, yielding an ASP-SL with low nitrogen content. Due to the distinctive carbon and nitrogen contents in the sludges, the C/N ratio also differed considerably between them. Maintaining total Kjeldahl nitrogen (N-TKN, %) and phosphorus content (P-P_2O_5, %) in the sewage sludge after the biological treatment would favor its valorization through the composting process for its subsequent merchandizing as fertilizer (Cao et al., 2021). Finally, the respirometric behavior of both types of sludge was indicative of the metabolic state of the microbial communities present in them. In this context, Table 3 depicts the microbiological activity of the sewage sludge, expressed in $SOUR_{max}$ (mg O_2/g VS·h), and its biodegradability, expressed as OD_{20} (mg O_2/ g VS). The $SOUR_{max}$ of both sewage sludges exhibited low values, although far from values that would suggest microbiological stability, that is, $SOUR_{max}$ < 1 mg O_2/g VS·h (Chica et al., 2003; Toledo et al., 2019). In a graphical manner, Fig. 1 shows the evolution of SOUR during the analysis, with a very different behavior at the beginning. In this sense, a marked decrease in SOUR was observed for ASP-SL,

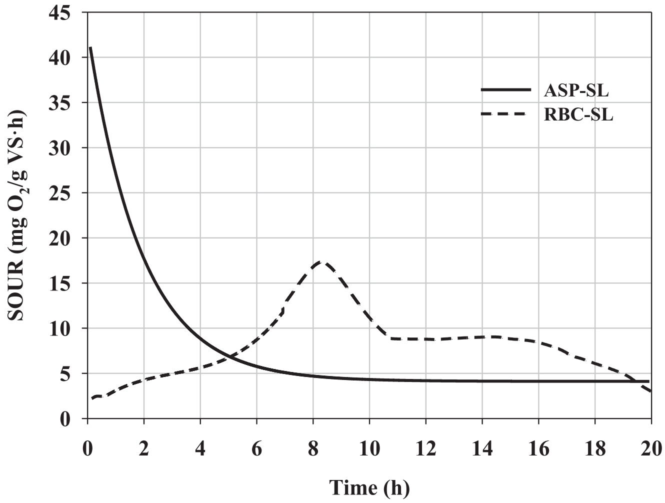


Fig. 1. Respirometric activity of sludge from ASP-WWTP and RBC-WWTP (ASP-SL and RBC-SL, respectively).

going from an initial maximum value of 41 ± 2 mg O_2/g VS·h to values lower than 5 mg O_2/g VS·h after 7 h of respirometric analysis. By contrast, the SOUR of RBC-SL increased slowly from the beginning until reaching its maximum value at approximately 8 h (17 ± 1 mg O_2/g VS·h). Afterwards, the above-mentioned parameter decreased rapidly until the 11 h (≈ 9 mg O_2/g VS·h) and continued to decrease slowly until the end of the analysis (≈ 3 mg O_2/g VS·h). Therefore, Fig. 1 shows that ASP-SL was in a more active metabolic state than RBC-SL, due to the endogenous respiration activity of microbes in the first sludge. It is well known that facilities operating with extended aeration, as a modification of the activated sludge process (such is the case of ASP-WWTP), give long HRTs to the aeration operation (≈ 14 h in ASP-WWTP). This extended aeration process is usually designed for complete-mix conditions, which means that high solids content or a low food-to-microorganism ratio and long sludge age will exist (see values of these parameters in Table S1). Thus, endogenous respiration of the sludge will occur and the sludge will "burn itself up" (Englande and Krenkel, 2003; Von Sperling, 2007). Nevertheless, regarding the cumulative oxygen demand, OD_{20} values indicated the high biodegradability of both sewage sludges, the biodegradability of RBC-SL being slightly higher (Table 3). It is important to note that the OD_{20} (mg O_2/ g VS) corresponded to the area under the curve for SOUR (mg O_2/g VS h) *versus* time (h).

The efficiency of a biological treatment could be determined by quantifying and characterizing the sewage sludge produced. The amount of generated sewage sludge and its physicochemical characteristics have a close relationship with the odorous impact generated by the wastewater treatment process. In this context, it is noteworthy that the amounts of sewage sludge (on a dry basis) generated by ASP-WWTP and RBC-WWTP were 149.19 g/EI·d and 72.08 g/EI·d, respectively. These results show that ASP is a more efficient technology in the biological treatment of wastewater, allowing a higher amount of sewage sludge per EI to be obtained as well as increasing the quality of the treated water (Table 2). Regarding the characteristics of the sewage sludge evaluated, Table 4 shows the nutrient content of both sludges generated by EI. Remarkably, ASP-SL was found to contain more organic matter (91.59 g VS/EI·d) than RBC-SL (55.26 g VS/EI·d), both expressed as VS. However, N-TKN and P-P_2O_5 showed slight variations, with ranges of similar magnitude in the two technologies. In the case of RBC-WWTP, N-TKN was maintained in the wastewater, avoiding its metabolization and emission as volatile compounds. As expected, ASP-SL was found to contain less nitrogen as ammoniacal nitrogen or in the soluble form (which are more

Table 4
Nutrient outlet flows (dry basis) with the sewage sludge from the WWTPs evaluated.

	ASP-SL	RBC-SL
C-VS (g/d·EI)	91.59	55.26
N-TKN (g/d·EI)	7.82	5.98
N-NH_4^+ (g/d·EI)	1.18	1.99
N-TN_s (g/d·EI)	0.64	1.83
P-P_2O_5 (g/d·EI)	8.07	3.55

ASP-SL, sludge from the wastewater treatment plant with activated sludge process; C-VS, organic matter expressed as volatile solids; EI, equivalent inhabitant; N-NH_4 $^+$, ammoniacal nitrogen; N-TKN, total Kjeldahl nitrogen; N-TNs, soluble total nitrogen; P-P2O5, phosphorus content expressed as P2O5; RBC-SL, sludge from the wastewater treatment plant with rotating biological contactors.

biodegradable forms), due to the nitrification-denitrification processes of the sewage during biological treatment.

3.3. Odor emission assessment

Having evaluated the wastewater and sludge characteristics, it was important to establish differences between the two WWTPs in terms of odor emission. In this sense, Table 5 shows the OCs and the odor emissions of each odor source sampled in both facilities. Regarding the preliminary treatment operations (pretreatment header and grit and grease removal), RBC-WWTP presented a higher OC in these stages, possibly due to the higher organic load received compared to ASP-WWTP. Moreover, regardless of the type of treatment technology used, the OCs associated with the first stages of the integral wastewater treatment were always found to be higher than the OCs emitted in the biological treatment of this water (Capelli et al., 2009). This important unit operation constituted a low-potential (and not unpleasant) odor source in both WWTPs, as is typical in most facilities of this type (Jiang et al., 2017). In this context, a higher OC was emitted in the biological treatment of ASP-WWTP, possibly owing to the fact that this technology favors a higher liquid-gas transfer (as a result of the extended aeration that takes place) compared to RBC technology, where biological disks rotate at very low speed, typically between 1 and 2 rpm (Cortez et al., 2008). Furthermore, ASP implies a greater transformation of organic compounds, as this technology has a higher performance in the elimination of organic pollutant load than RBC. The presence of the anoxic reactor in ASP-WWTP also contributed to increase the OC emitted at the biological treatment stage in this facility, since VSC emissions can be increased during the anoxic cycle (Kim et al., 2014). After the biological treatment, the OC emitted increased again for both WWTPs in the secondary settling step (Table 5), potentially related to the formation and release of volatile organic sulfur compounds due to the anoxic environment in settled sludge (Sekyiamah et al., 2008).

On the other hand, the sludge handling activities (sludge thickening and dewatering) constituted the major sources of odor emissions regardless of the WWTP evaluated (99.46% and 99.80% in ASP-WWTP and RBC-WWTP, respectively). The high odor potential of the aforementioned activities has been widely reported in the literature, this being mainly due to the acidic and anaerobic conditions that exist at these steps, leading to the formation of H_2S and malodorous organic compounds, mainly VSCs (Karageorgos et al., 2010; Lebrero et al., 2011; Márquez et al., 2021). Nevertheless, as can be observed in Table 5, RBC-WWTP presented higher emitted OCs than those of ASP-WWTP at the sludge thickening and dewatering stages. This could be due to the fact that RBC-SL was less mineralized than ASP-SL, given its higher organic matter content (76.66% *vs.* 61.69% of VS, respectively). In line with the above, a positive relationship between the organic matter content of different wastes and its odor impact is

Table 5
Odor emissions from the WWTPs.

	Odor source	OC (ou_E/m^3)	SOER ($ou_E/m^2{\cdot}s$)	Emission surface (m^2)	OER (ou_E/s)	Global odor emission (ou_E/s)
ASP-WWTP	Pretreatment header	108	0.22	20.00	4.49	11,177
	Grit and grease removal	108	0.22	29.75	6.68	
	Biological treatment (aerobic reactor)	76	0.16	68.00	10.79	
	Biological treatment (anoxic reactor)	93	0.19	22.50	4.35	
	Secondary settling	128	0.27	127.23	33.89	
	Sludge thickening	2030	897.46	7.00	6282.20	
	Sludge dewatering	1024	452.66	10.68	4834.40	
RBC-WWTP	Pretreatment header	256	0.53	1.80	0.96	12,784
	Grit and grease removal	197	0.41	6.31	2.59	
	Biological treatment	41	0.09	21.68	1.87	
	Secondary settling	152	0.32	63.62	20.14	
	Sludge thickening	2903	1283.41	3.00	3850,22	
	Sludge dewatering	2048	905.36	9.84	8908.71	

ASP-WWTP, wastewater treatment plant with activated sludge process; OC, odor concentration; OER, odor emission rate; RBC-WWTP, wastewater treatment plant with rotating biological contactors; SOER, specific odor emission rate.

Table 6
Studies on odor emissions depending on the wastewater treatment technology.

Wastewater treatment technology	What is evaluated?	How is it evaluated?	Reference
ASP (extended aeration)	Odor, H_2S and NH_3 emissions	Olfactometry analysis and air quality measurements of H_2S and NH_3	Dinçer et al. (2020)
ASP (extended aeration)	H_2S emission	Estimated H_2S through TOXCHEM model (it is based on mass balance of several compounds in WWTPs for each operation unit)	Zwain et al. (2020)
ASP	VSCs emission	Real-time total reduced sulfur analyser	Kim et al. (2014)
ASP	Odor emission	OEFs calculated from measurements using dynamic olfactometry	Varela-Bruce and Antileo (2021)
RBC	H_2S emission	Appearance of H_2S emission due to organic overload	Cortez et al. (2008)
SBR	VSCs emission	Chemical analysis (GC-FPD), odor active values and emission factors	Li et al. (2021)
ASP (extended aeration) *versus* RBC	Odor emission	Olfactometry analysis	The present study

ASP, activated sludge process; GC-FPD, gas chromatography – flame photometric detector; OEFs, odor emission factors; RBC, rotating biological contactors; SBR, sequencing batch reactor; VSCs, volatile sulfur compounds.

widely reported in the literature (Dunlop et al., 2016; Toledo et al., 2018a).

Regarding the global odor emissions, and considering that olfactometric values have a deviation between half and double (EN 13725, 2003), it might be said that ASP-WWTP and RBC-WWTP had a similar global odor emission rate, although this parameter is slightly higher in the second facility (Table 5). However, when considering odor emissions per equivalent inhabitant (that is, a specific odor emission rate with units of $ou_E/s{\cdot}EI$), a higher value was obtained for ASP-WWTP (16.22 $ou_E/s{\cdot}EI$) compared to RBC-WWTP (6.84 $ou_E/s{\cdot}EI$). Therefore, it can be said that RBC-WWTP would generate lower odor emissions than ASP-WWTP for the same organic load to be treated. In other words, regardless of the influent organic load, lower values of specific odor emission rate ($ou_E/s{\cdot}EI$) indicate a lower odor impact on the environment. At this point, it is important to note that the higher odor emission per EI associated with ASP-WWTP could have been related to the lower biological stability of the sludge generated (ASP-SL), since its microbiological activity, expressed in terms of specific oxygen uptake rate (SOUR, mg O_2/g VS·h), was up to 2.41 times higher than in RBC-SL (Table 3 and Fig. 1). Consistent with the above, a positive correlation was found in the literature between the emitted OC and SOUR in other biological processes, such as composting, where odor emission tends to decrease as the biological stability of the material increases (González et al., 2019; Gutiérrez et al., 2014; Toledo et al., 2018c).

Therefore, in small and medium-sized WWTPs (such is the case under study), the biological treatment technology used plays a key role in their odor impact, as it conditions sludge characteristics in subsequent sludge handling activities, which are the major sources of odor emission. Moreover, this study shows that RBC-WWTP would generate a lower odor impact than ASP-WWTP (based on extended aeration) for the treatment of the same organic load. This information might help to select the most appropriate biological wastewater treatment for minimizing the impact of odors when it is necessary to build new WWTPs, with the consequent social and environmental benefits.

By way of example, Table 6 shows how other authors have addressed the problem of odor emission in WWTPs with different biological treatment technologies. As can be observed in this table, most of the studies have focused on WWTPs with activated sludge process in its different modifications (Dinçer et al., 2020; Kim et al., 2014; Varela-Bruce and Antileo, 2021; Zwain et al., 2020). The odor impact of the technology based on rotating biological contactors has been much less studied and based solely on the emission of H_2S (Cortez et al., 2008). On the other hand, although the implementation of sequencing batch reactors is not very widespread in full-scale sewage treatment plants, Li et al. (2021) have characterized VSC emissions from a SBR-WWTP at the water-air interface. However, the present study goes a step further because, to date, it is the only one that performs a comparative evaluation of the odor emissions derived from two full-scale WWTPs with ASP (based on extended aeration) and RBC technologies, widely implemented in facilities of small and medium-sized municipalities (Sikosana et al., 2019; Wee Seow et al., 2016).

3.4. Bacterial microbiome analysis

Globally, the 16S sequencing analysis identified over 370,000 bacteria from the two WWTP sludges and 3 biological replicates analysed in this study. RBC-SL showed a higher number of identifications than ASP-SL, with almost 215,000 *versus* 160,000 counts, respectively. The difference in the number of identified bacteria in these sludges was in accordance with the higher VS concentration in RBC-SL than ASP-SL. The endogenous respiration of ASP-SL reduced

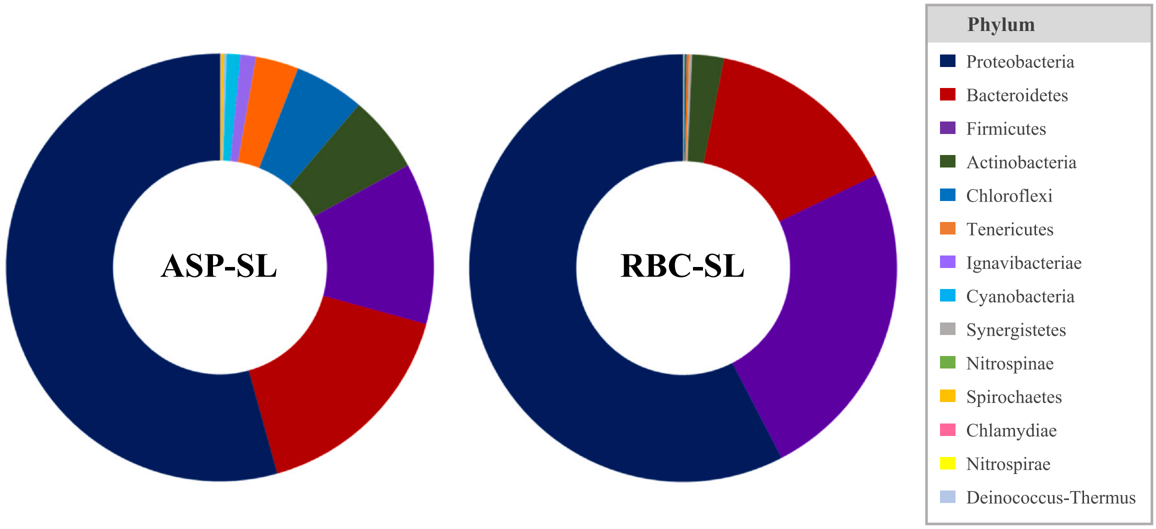


Fig. 2. Relative bacterial community composition at the phylum level, obtained after massive 16S rRNA sequencing of sludge from ASP-WWTP and RBC-WWTP (ASP-SL and RBC-SL, respectively). The mean of the 3 biological replicates per location is represented.

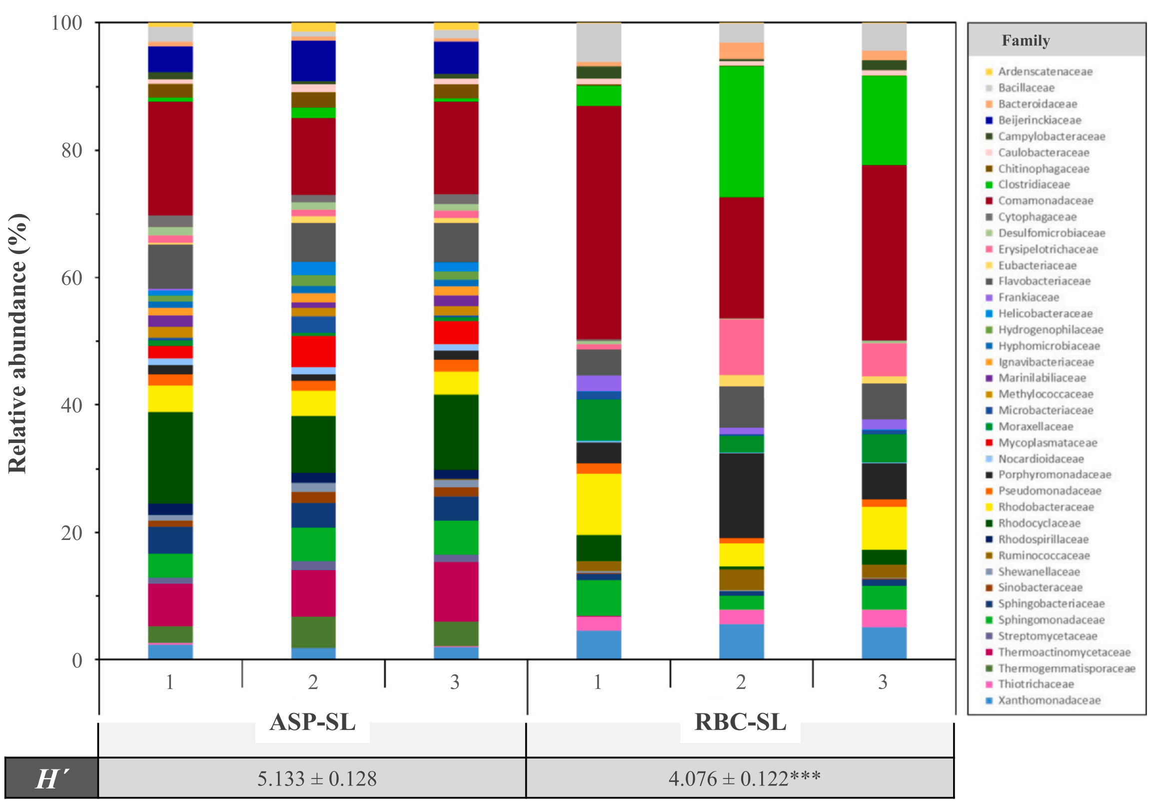


Fig. 3. Relative abundances of bacterial families and estimation of microbial biodiversity in ASP-SL and RBC-SL. The 3 biological replicates for each location are represented individually. Alpha diversity values are shown in terms of richness using the Shannon–Wiener index (H′). H´max was 7.15.

the concentration of biodegradable organic matter and thus the number of identified bacteria in this sludge.

Eleven and 13 different phyla were identified in ASP-SL and RBC-SL, respectively (Fig. 2). Proteobacteria, Bacteroidetes and Firmicutes were the three predominant phyla in both WWTPs, albeit with some differences. Proteobacteria represented the most common phylum in both, reaching a similar percentage (≈55%). Bacteroidetes also presented similar levels in both WWTPs (≈16%), although whereas they were the second-most abundant in ASP-SL, they were only the third-most abundant in RBC-SL due to the presence of a higher proportion of Firmicutes (25.6%). In ASP-SL, the three predominant phyla accounted for over 83% of all identifications, with 3 additional phyla ranging from 3.1% to 5.7% (Actinobacteria, Chloroflexi and Tenericutes), while the rest were infrequent (less than 1.2%). On the other hand, the three predominant phyla in RBC-SL collectively represented 97%, followed by Actinobacteria (2.3%), while the other 9 phyla identified were very scarce (less than 0.24%). Bacteria belonging to the Ignavibacteriae phylum were only identified in ASP-SL and those belonging to the Chlamydiae, Nitrospinae and Spirochaetes phyla were exclusive to RBC-SL (Fig. 2).

At the family level, the profiles of both sludges differed in accordance with the different biological processes analysed (Fig. 3 and Table S2, Supplementary material). The high degree of similarity among the biological replicate profiles in both sludges should be acknowledged. Among the 142 families identified, 90 were present in both WWTPs (63%), while 35 and 17 were only detected in ASP-SL and RBC-SL, respectively. This imbalance was also quantified using the Shannon–Wiener index (H′), which estimates microbial biodiversity in a specific environment using the abundance of species against their relative evenness. As can be observed in Fig. 3, the biodiversity of the bacterial microbiota in RBC-SL was significatively lower than in ASP-SL, according to the lower number of families detected in the first sludge. These data are in line with previous reports indicating that ASP sludges present highly diverse microbial ecosystems (Wu et al., 2019). The presence of pollutants has been previously associated with biodiversity reduction and thus with more vulnerable aquatic ecosystems (Johnston and Roberts, 2009). Although nutrient enrichment can have positive effects on species richness (Johnston and Roberts, 2009), the pressure of a high organic pollutant load in RBC-WWTP contributed to its reduced bacterial diversity. The Proteobacteria Comamonadaceae was the most common family in both sludges, representing 13.2% and 25.6% in ASP-SL and RBC-SL, respectively. Fifty-two families presented significantly different counts between the two WWTP sludges (Table S2, Supplementary material). Although RBC-SL presented a higher global number of identifications, most of the families with significant differences (77%) had higher levels in ASP-SL, being Beijerinckiaceae up to 254 higher than in RBC-SL. Only 12 families (23%) were more abundant in RBC-SL, most notably Ruminococcacea which was 71 times more common than in ASP-SL.

Most studies of the microbiomes in WWTPs are focused on ASP-based facilities. Recently, a very comprehensive study of the bacterial composition of ASP sludges from a wide range of locations across the world proposed a core bacterial community, highlighting certain genera that were present in most samples, for instance *Dokdonella*, *Zoogloea*, *Nitrospira* and *Arcobacter* (Wu et al., 2019). On the other hand, studies on the microbial composition of RBC-based facilities remain scarce, although several operational taxonomic units (OTUs) have been identified and related to nitrification, ammonia-oxidizing and denitrification processes in these reactors, *e.g. Nitrospira* , *Nitrosomonas*, *Paracoccus*, *Rhodobacter* and *Thauera* (Peng et al., 2014; Spasov et al., 2020). All these genera were also identified in our samples, but no significant differences in their abundance were found between the ASP-SL and RBC-SL samples (data not shown).

Sludge contains a lot of anaerobic organisms, which produce odors when they consume organic matter (Holman and Wareham, 2003). Thus, odors are usually the result of the activity of microbiota, as by consuming organic matter, they produce VOCs and odorous gases such as hydrogen sulfide or ammonia. In concordance with the higher global odor emission in RBC-WWTP compared to ASP-WWTP, several microorganisms that the present study found to be significantly higher in RBC-SL have previously been linked to the production of unpleasant odors. In this sense, Porphyromonadaceae, Clostridiales, Lachnospiraceae and Moraxellaceae are some examples (see Table S2, Supplementary material), all members of the first three families being obligate anaerobes (Hummelen et al., 2010; Kubota et al., 2012; Kumpitsch et al., 2019; Million et al., 2020). Therefore, this fact is in line with the higher OC from the sludge handling activities (sludge thickening and dewatering) in RBC-WWTP compared to ASP-WWTP (Table 5). However, it is also worth noting that sulfur-oxidizing bacteria, that are able to consume some of these odor compounds and thus participate in nuisance odor removal (Fan et al., 2020; Ren et al., 2019), were found to be higher in RBC-SL than in ASP-SL, specifically Halothiobacillaceae (12.9-fold), Thiotrichaceae (32.2-fold) and Xanthomonadaceae (3.5-fold).

In WWTPs with rotating biological contactors it is difficult to minimize odor emissions due to the presence of anaerobic metabolic pathways that depend on both the influent organic load and the thickness of the biodiscs´ biofilm (Cortez et al., 2008). In this context, the use of closed systems for the sludge handling activities that are equipped with a deodorization system based on GAC beds could be the most appropriate technology for efficient odor removal in small and medium-sized WWTPs operating with RBC. However, the consumption of resources in the facility would increase notably (especially in terms of energy and adsorbent material). For its part, WWTPs with activated sludge process have a better responsiveness to increases in influent organic load (that could generate higher odor emissions) than RBC-WWTPs, due to the possibility of increasing the solids concentration in the reactor. Nevertheless, increasing the concentration of solids also has some disadvantages, because as such concentration increases the efficiency of aeration decreases. In this sense, if the oxygenation capacity is not sufficient, the organic overload could even lead to anaerobic conditions and, therefore, higher odor emissions (Von Sperling, 2007). Thus, maintaining the solids concentration at optimal levels in the reactor is critical for both wastewater treatment performance and odor management in ASP-WWTPs. However, it would be also advisable to implement a deodorization system (such as the one previously mentioned) to control the odor emissions derived from the sludge handling activities, which also constitute the major odor emission sources in small and medium-sized ASP-WWTPs based on extended aeration.

4. Conclusions

The WWTP with activated sludge process based on extended aeration had a global odor emission very close to that of the WWTP with rotating biological contactors (11,177 and 12,784 ou_E/s, respectively), although ASP-WWTP received an organic load approximately 2.7 times lower than that of RBC-WWTP. The sludge handling activities constituted the major odor emission sources in both facilities (accounting for 99.46% and 99.80% of the global odor emission in ASP-WWTP and RBC-WWTP, respectively). The significative increments in the abundance of anaerobic microbial families such as Porphyromonadaceae, Clostridiales, Lachnospiraceae and the aerobic Moraxellaceae in RBC-SL helped to justify the higher odor concentration from the sludge handling activities in RBC-WWTP compared to ASP-WWTP. However, regardless of the influent organic load, ASP-WWTP would have a higher specific odor emission than RBC-WWTP (16.22 and 6.84 $ou_E/s{\cdot}EI$, respectively), which is related to the different respirometric activity of the sludge generated. In this sense, the maximum specific oxygen uptake rate (mg O_2/g VS·h) was 2.41 times higher in ASP-SL than RBC-SL.

Given the above, this study provides relevant information about the odor emission of small and medium-sized full-scale WWTPs, which may help wastewater managers to select the most appropriate biological wastewater treatment technology from an environmental and human health point of view, since it conditions sludge characteristics, and therefore, the odor impact of those facilities.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.psep.2022.02.071.

References

Aonofriesei, F., Petrosanu, M., 2007. Activated sludge bulking episodes and dominant filamentous bacteria at wastewater treatment plant Constanţa Sud (Romania). Microbiology 2, 83–87.
APHA, 2017. Standard methods for the examination of water and wastewater, 23rd ed. American Public Health Association, Washington DC.
Bhatla, M.N., 1975. Control of odors at an activated sludge plant. Water Pollut. Control Fed. 47, 281–290.
Byliński, H., Gębicki, J., Namieśnik, J., 2019. Evaluation of health hazard due to emission of volatile organic compounds from various processing units of wastewater treatment plant. Int. J. Environ. Res. Public Health 16. https://doi.org/10.3390/ijerph16101712
Cao, L., Liao, L., Su, C., Mo, T., Zhu, F., Qin, R., Li, R., 2021. Metagenomic analysis revealed the microbiota and metabolic function during co-composting of food waste and residual sludge for nitrogen and phosphorus transformation. Sci. Total Environ. 773, 145561. https://doi.org/10.1016/j.scitotenv.2021.145561
Capelli, L., Sironi, S., Del Rosso, R., Céntola, P., 2009. Predicting odour emissions from wastewater treatment plants by means of odour emission factors. Water Res. 43, 1977–1985. https://doi.org/10.1016/j.watres.2009.01.022
Carrera-Chapela, F., Donoso-Bravo, A., Souto, J.A., Ruiz-Filippi, G., 2014. Modeling the odor generation in WWTP: an integrated approach review. Water Air Soil Pollut. 225, 1–15. https://doi.org/10.1007/S11270-014-1932-Y
Chahal, C., van den Akker, B., Young, F., Franco, C., Blackbeard, J., Monis, P., 2016. Pathogen and particle associations in wastewater: significance and implications for treatment and disinfection processes. Adv. Appl. Microbiol. 97, 63–119. https://doi.org/10.1016/BS.AAMBS.2016.08.001
Chica, A., Mohedo, J.J., Martín, M.A., Martín, A., 2003. Determination of the stability of msw compost using a respirometric technique. Compost Sci. Util. 11, 169–175. https://doi.org/10.1080/1065657X.2003.10702122
Cortez, S., Teixeira, P., Oliveira, R., Mota, M., 2008. Rotating biological contactors: a review on main factors affecting performance. Rev. Environ. Sci. Biotechnol. 7, 155–172. https://doi.org/10.1007/s11157-008-9127-x
Dinçer, F., Dinçer, F.K., Sarı, D., Ceylan, Ö., Ercan, Ö., 2020. Dispersion modeling and air quality measurements to evaluate the odor impact of a wastewater treatment plant in İzmir. Atmos. Pollut. Res. 11, 2119–2125. https://doi.org/10.1016/J.APR.2020.05.018
Dominguez, D., Gujer, W., 2006. Evolution of a wastewater treatment plant challenges traditional design concepts. Water Res. 40, 1389–1396. https://doi.org/10.1016/j.watres.2006.01.034
Dong, D., Yan, A., Liu, H., Zhang, X., Xu, Y., 2006. Removal of humic substances from soil DNA using aluminium sulfate. J. Microbiol. Methods 66, 217–222. https://doi.org/10.1016/j.mimet.2005.11.010
Dunlop, M., Blackall, P., Stuetz, R., 2016. Odour emissions from poultry litter – a review litter properties, odour formation and odorant emissions from porous materials. J. Environ. Manag. 177, 306–319. https://doi.org/10.1016/J.JENVMAN.2016.04.009
EN 13725 2003. Air Quality - Determination of Odour Concentration by Dynamic Olfactometry. European Committee for Standardization, Brussels.
Englande, A.J., Krenkel, P.A., 2003. Waste water treatment and water reclamation. In: Encyclopedia of Physical Science and Technology. Academic Press, pp. 639–670. https://doi.org/10.1016/B0-12-227410-5/00818-8
EU Council, 1991. Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment.
Fan, F., Xu, R., Wang, D., Meng, F., 2020. Application of activated sludge for odor control in wastewater treatment plants: approaches, advances and outlooks. Water Res. 181, 115915. https://doi.org/10.1016/j.watres.2020.115915
Ferrer, C., Miguel, D., Ferrer, L., Alonso, S., Sangüesa, I., Basiero, A., Bernacer, I., Morenilla, J.J., 2008. Análisis comparativo entre procesos de tratamiento de agua residual de pequeñas poblaciones de la provincia de Castellón. Tecnol. Del. Agua 28, 30–36.
García, L., Leyva-Díaz, J.C., Díaz, E., Ordóñez, S., 2021. A review of the adsorption-biological hybrid processes for the abatement of emerging pollutants: removal efficiencies, physicochemical analysis, and economic evaluation. Sci. Total Environ. 780, 146554. https://doi.org/10.1016/J.SCITOTENV.2021.146554
González, D., Colón, J., Sánchez, A., Gabriel, D., 2019. A systematic study on the VOCs characterization and odour emissions in a full-scale sewage sludge composting plant. J. Hazard. Mater. 373, 733–740. https://doi.org/10.1016/J.JHAZMAT.2019.03.131
Gutiérrez, M.C., Serrano, A., Martín, M.A., Chica, A.F., 2014. Odour in composting processes at pilot scale: monitoring and biofiltration. Environ. Technol. 35, 1676–1684. https://doi.org/10.1080/09593330.2014.880132
Hassard, F., Biddle, J., Cartmell, E., Jefferson, B., Tyrrel, S., Stephenson, T., 2015. Rotating biological contactors for wastewater treatment – a review. Process Saf. Environ. Prot. 94, 285–306. https://doi.org/10.1016/J.PSEP.2014.07.003
Holman, J.B., Wareham, D.G., 2003. Oxidation-reduction potential as a monitoring tool in a low dissolved oxygen wastewater treatment process. J. Environ. Eng. 129, 52–58. https://doi.org/10.1061/(ASCE)0733-9372(2003)129:1(52)
Hummelen, R., Fernandes, A.D., Macklaim, J.M., Dickson, R.J., Changalucha, J., Gloor, G.B., Reid, G., 2010. Deep sequencing of the vaginal microbiota of women with HIV. PLOS One 5. https://doi.org/10.1371/journal.pone.0012078
Jácome, A., Suárez, J., Ures, P., 2015. Rotating biological contactors (biodiscs) (FS-BIO-004) [Fact sheet] [WWW Document]. URL ⟨https://www.wateractionplan.com/documents/177327/558166/Rotating+biological+contactors.pdf⟩ (accessed 5.25.21).
Jiang, G., Melder, D., Keller, J., Yuan, Z., 2017. Odor emissions from domestic wastewater: a review. Crit. Rev. Environ. Sci. Technol. 47, 1581–1611. https://doi.org/10.1080/10643389.2017.1386952
Johnston, E.L., Roberts, D.A., 2009. Contaminants reduce the richness and evenness of marine communities: a review and meta-analysis. Environ. Pollut. 157, 1745–1752. https://doi.org/10.1016/j.envpol.2009.02.017
Kalbar, P.P., Karmakar, S., Asolekar, S.R., 2012. Technology assessment for wastewater treatment using multiple-attribute decision-making. Technol. Soc. 34, 295–302. https://doi.org/10.1016/j.techsoc.2012.10.001
Karageorgos, P., Latos, M., Kotsifaki, C., Lazaridis, M., Kalogerakis, N., 2010. Treatment of unpleasant odors in municipal wastewater treatment plants. Water Sci. Technol. 61, 2635–2644. https://doi.org/10.2166/wst.2010.211
Kim, H., Lee, H., Choi, E., Choi, I., Shin, T., Im, H., Ahn, S., 2014. Characterization of odor emission from alternating aerobic and anoxic activated sludge systems using real-time total reduced sulfur analyzer. Chemosphere 117, 394–401. https://doi.org/10.1016/j.chemosphere.2014.08.008
Kominko, H., Gorazda, K., Wzorek, Z., 2019. Potentiality of sewage sludge-based organo-mineral fertilizer production in Poland considering nutrient value, heavy metal content and phytotoxicity for rapeseed crops. J. Environ. Manag. 248, 109283. https://doi.org/10.1016/j.jenvman.2019.109283
Kubota, H., Mitani, A., Niwano, Y., Takeuchi, K., Tanaka, A., Yamaguchi, N., Kawamura, Y., Hitomi, J., 2012. Moraxella species are primarily responsible for generating malodor in laundry. Appl. Environ. Microbiol. 78, 3317–3324. https://doi.org/10.1128/AEM.07816-11
Kumpitsch, C., Koskinen, K., Schöpf, V., Moissl-Eichinger, C., 2019. The microbiome of the upper respiratory tract in health and disease. BMC Biol. 17, 1–20. https://doi.org/10.1186/s12915-019-0703-z
Lebrero, R., Bouchy, L., Stuetz, R., Muñoz, R., 2011. Odor assessment and management in wastewater treatment plants: a review. Crit. Rev. Environ. Sci. Technol. 41, 915–950. https://doi.org/10.1080/10643380903300000
Li, R., Han, Z., Shen, H., Qi, F., Ding, M., Song, C., Sun, D., 2021. Emission characteristics of odorous volatile sulfur compound from a full-scale sequencing batch reactor wastewater treatment plant. Sci. Total Environ. 776, 145991. https://doi.org/10.1016/j.scitotenv.2021.145991
Liu, Y., Ngo, H.H., Guo, W., Peng, L., Wang, D., Ni, B., 2019. The roles of free ammonia (FA) in biological wastewater treatment processes: a review. Environ. Int 123, 10–19. https://doi.org/10.1016/j.envint.2018.11.039
Márquez, P., Benítez, A., Caballero, Á., Siles, J.A., Martín, M.A., 2021. Integral evaluation of granular activated carbon at four stages of a full-scale WWTP deodorization system. Sci. Total Environ. 754, 142237. https://doi.org/10.1016/j.scitotenv.2020.142237
Martín, M.A., Gutiérrez, M.C., Dios, M., Siles, J.A., Chica, A.F., 2018. Application of ATAD technology for digesting sewage sludge in small towns: operation and costs. J. Environ. Manag. 215, 185–194. https://doi.org/10.1016/J.JENVMAN.2018.03.062
Million, M., Armstrong, N., Khelaifia, S., Guilhot, E., Richez, M., Lagier, J.C., Dubourg, G., Chabriere, E., Raoult, D., 2020. The antioxidants glutathione, ascorbic acid and uric acid maintain butyrate production by human gut clostridia in the presence of oxygen in vitro. Sci. Rep. 10, 1–11. https://doi.org/10.1038/s41598-020-64834-3
Oh, S.T., Kim, J.R., Premier, G.C., Lee, T.H., Kim, C., Sloan, W.T., 2010. Sustainable wastewater treatment: how might microbial fuel cells contribute. Biotechnol. Adv. 28, 871–881. https://doi.org/10.1016/j.biotechadv.2010.07.008
Omri, I., Bouallagui, H., Aouidi, F., Godon, J.J., Hamdi, M., 2011. H2S gas biological removal efficiency and bacterial community diversity in biofilter treating wastewater odor. Bioresour. Technol. 102, 10202–10209. https://doi.org/10.1016/j.biortech.2011.05.094
Peng, X., Guo, F., Ju, F., Zhang, T., 2014. Shifts in the microbial community, nitrifiers and denitrifiers in the biofilm in a full-scale rotating biological contactor. Environ. Sci. Technol. 48, 8044–8052. https://doi.org/10.1021/es5017087
Prasse, C., Stalter, D., Schulte-Oehlmann, U., Oehlmann, J., Ternes, T.A., 2015. Spoilt for choice: a critical review on the chemical and biological assessment of current wastewater treatment technologies. Water Res. 87, 237–270. https://doi.org/10.1016/j.watres.2015.09.023
Rana, S., Gupta, N., Rana, R.S., 2018. Removal of organic pollutant with the use of rotating biological contactor. In: Materials Today: Proceedings. Elsevier Ltd, pp. 4218–4224. https://doi.org/10.1016/j.matpr.2017.11.685
Ren, B., Zhao, Y., Lyczko, N., Nzihou, A., 2019. Current status and outlook of odor removal technologies in wastewater treatment plant. Waste Biomass Valoriz. 10, 1443–1458. https://doi.org/10.1007/s12649-018-0384-9
Saeed, M.U., Hussain, N., Sumrin, A., Shahbaz, A., Noor, S., Bilal, M., Aleya, L., Iqbal, H.M.N., 2021. Microbial bioremediation strategies with wastewater treatment potentialities – a review. Sci. Total Environ., 151754. https://doi.org/10.1016/J.SCITOTENV.2021.151754

Sekyiamah, K., Kim, H., McConnell, L.L., Torrents, A., Ramirez, M., 2008. Identification of seasonal variations in volatile sulfur compound formation and release from the secondary treatment system at a large wastewater treatment plant. Water Environ. Res. 80, 2261–2267. https://doi.org/10.2175/106143008×304677
Sikosana, M.L., Sikhwivhilu, K., Moutloali, R., Madyira, D.M., 2019. Municipal wastewater treatment technologies: a review. Procedia Manuf. 35, 1018–1024. https://doi.org/10.1016/j.promfg.2019.06.051
Spasov, E., Tsuji, J.M., Hug, L.A., Doxey, A.C., Sauder, L.A., Parker, W.J., Neufeld, J.D., 2020. High functional diversity among Nitrospira populations that dominate rotating biological contactor microbial communities in a municipal wastewater treatment plant. ISME J. 14, 1857–1872. https://doi.org/10.1038/s41396-020-0650-2
Tchobanoglous, G., Burton, F.L., Stensel, H.D., 2003. Wastewater Engineering: Treatment and Reuse. McGraw Hill, New York, NY.
Toledo, M., Gutiérrez, M.C., Siles, J.A., García-Olmo, J., Martín, M.A., 2018a. Chemometric analysis and NIR spectroscopy to evaluate odorous impact during the composting of different raw materials. J. Clean. Prod. 167, 154–162. https://doi.org/10.1016/j.jclepro.2017.08.163
Toledo, M., Siles, J.A., Gutiérrez, M.C., Martín, M.A., 2018c. Monitoring of the composting process of different agroindustrial waste: Influence of the operational variables on the odorous impact. Waste Manag. 76, 266–274. https://doi.org/10.1016/j.wasman.2018.03.042
Toledo, M., Márquez, P., Siles, J.A., Chica, A.F., Martín, M.A., 2019. Co-composting of sewage sludge and eggplant waste at full scale: feasibility study to valorize eggplant waste and minimize the odoriferous impact of sewage sludge. J. Environ. Manag. 247, 205–213. https://doi.org/10.1016/j.jenvman.2019.06.076
Ullah, A., Hussain, S., Wasim, A., Jahanzaib, M., 2020. Development of a decision support system for the selection of wastewater treatment technologies. Sci. Total Environ. 731, 139158. https://doi.org/10.1016/j.scitotenv.2020.139158
US Department of Agriculture and the US Composting Council, 2002. Test Methods for the Examination of Composting and Compost (TMECC). Edaphos International, Houston, TX.
Varela-Bruce, C., Antileo, C., 2021. Assessment of odour emissions by the use of a dispersion model in the context of the proposed new law in Chile. J. Environ. Manag. 295, 113208. https://doi.org/10.1016/J.JENVMAN.2021.113208
VDI guideline 3880, 2011. Olfactometry – Static Sampling. Beuth Verlag GmbH, Berlin, pp. 10772.
Von Sperling, M., 2007. Basic Principles of Wastewater Treatment. IWA Publishing. https://doi.org/10.2166/9781780402093.
Wang, Y., Jin, X., He, L., Zhang, W., 2017. Inhibitory effect of thiourea on biological nitrification process and its eliminating method. Water Sci. Technol. 75, 2900–2907. https://doi.org/10.2166/WST.2017.177
Waqas, S., Bilad, M.R., 2019. A review on rotating biological contactors. Indones. J. Sci. Technol. 4, 241–256. https://doi.org/10.17509/IJOST.V4I2.18181
Waqas, S., Bilad, M.R., Man, Z., Wibisono, Y., Jaafar, J., Indra Mahlia, T.M., Khan, A.L., Aslam, M., 2020. Recent progress in integrated fixed-film activated sludge process for wastewater treatment: a review. J. Environ. Manag. 268, 110718. https://doi.org/10.1016/j.jenvman.2020.110718
Wee Seow, T., Lim, C.K., Hanif, M., Nor, M., Fahmi, M., Mubarak, M., Lam, C.Y., Yahya, A., Ibrahim, Z., 2016. Review on wastewater treatment technologies. Int. J. Appl. Environ. Sci. 11, 111–126.
Wei, W., Wang, Q., Zhang, L., Laloo, A., Duan, H., Batstone, D.J., Yuan, Z., 2018. Free nitrous acid pre-treatment of waste activated sludge enhances volatile solids destruction and improves sludge dewaterability in continuous anaerobic digestion. Water Res. 130, 13–19. https://doi.org/10.1016/j.watres.2017.11.050
Wu, L., Ning, D., Zhang, B., Li, Y., Zhang, P., Shan, X., Zhang, Q., Brown, M., Li, Z., Van Nostrand, J.D., Ling, F., Xiao, N., Zhang, Y., Vierheilig, J., Wells, G.F., Yang, Y., Deng, Y., Tu, Q., Wang, A., Zhang, T., He, Z., Keller, J., Nielsen, P.H., Alvarez, P.J.J., Criddle, C.S., Wagner, M., Tiedje, J.M., He, Q., Curtis, T.P., Stahl, D.A., Alvarez-Cohen, L., Rittmann, B.E., Wen, X., Zhou, J., Acevedo, D., Agullo-Barcelo, M., Andersen, G.L., de Araujo, J.C., Boehnke, K., Bond, P., Bott, C.B., Bovio, P., Brewster, R.K., Bux, F., Cabezas, A., Cabrol, L., Chen, S., Etchebehere, C., Ford, A., Frigon, D., GÃ3mez, J.S., Griffin, J.S., Gu, A.Z., Habagil, M., Hale, L., Hardeman, S.D., Harmon, M., Horn, H., Hu, Z., Jauffur, S., Johnson, D.R., Kumari A., Keucken, Leal, S., Lebrun, C.D., L.A, Lee, J., Lee, M., Lee, Z.M.P., Li, M., Li, X., Liu, Y., Luthy, R.G., MendonÃ§a-Hagler, L.C., de Menezes, F.G.R., Meyers, A.J., Mohebbi, A., Oehmen, A., Palmer, A., Parameswaran, P., Park, J., Patsch, D., Reginatto, V., de los Reyes, F.L., Noyola, A., Rossetti, S., Sidhu, J., Sloan, W.T., Smith, K., de Sousa, O.V., Stephens, K., Tian, R., Tooker, N.B., De los Cobos D., Vasconcelos, Wakelin, S., Wang, B., Weaver, J.E., West, S., Wilmes, P., Woo, S.G., Wu, J.H., Wu, L., Xi, C., Xu, M., Yan, T., Yang, M., Young, M., Yue, H., Zhang, Q., Zhang, W., Zhang, Y., Zhou, H., 2019. Global diversity and biogeography of bacterial communities in wastewater treatment plants. Nat. Microbiol. 4, 1183–1195. https://doi.org/10.1038/s41564-019-0426-5
Yu, K., Zhang, T., 2012. Metagenomic and metatranscriptomic analysis of microbial community structure and gene expression of activated sludge. PLOS One 7, e38183. https://doi.org/10.1371/journal.pone.0038183
Zhang, H., Zheng, S., Zhang, X., Duan, S., Li, S., 2020. Optimizing the inclined plate settler for a high-rate microaerobic activated sludge process for domestic wastewater treatment: a theoretical model and experimental validation. Int. Biodeterior. Biodegrad. 154, 105060. https://doi.org/10.1016/j.ibiod.2020.105060
Ziembińska-Buczyńska, A., Ciesielski, S., Żabczyński, S., Cema, G., 2019. Bacterial community structure in rotating biological contactor treating coke wastewater in relation to medium composition. Environ. Sci. Pollut. Res. 26, 19171–19179. https://doi.org/10.1007/s11356-019-05087-0
Zwain, H.M., Nile, B.K., Faris, A.M., Vakili, M., Dahlan, I., 2020. Modelling of hydrogen sulfide fate and emissions in extended aeration sewage treatment plant using TOXCHEM simulations. Sci. Rep. 101 (10), 1–11. https://doi.org/10.1038/s41598-020-79395-8