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Review

Odour prevention strategies in wastewater treatment and composting plants: A review

M. Toledo [a,b,**], R. Muñoz [a,b,*]

[a] Institute of Sustainable Processes, Dr. Mergelina s/n, 47011, Valladolid, Spain
[b] Department of Chemical Engineering and Environmental Technology, School of Industrial Engineering, University of Valladolid, Dr. Mergelina, s/n, 47011, Valladolid, Spain

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ABSTRACT

Odour emissions from wastewater treatment plants (WWTPs) and composting plants (CPs) have become a critical environmental and public health issue due to the release of complex mixtures of volatile organic compounds, volatile inorganic compounds, and volatile sulphur compounds. These emissions do not only affect ambient air quality but also contribute to nuisance complaints and potential health risks in the nearby communities. This paper provides a comprehensive review of current odour prevention strategies employed in WWTPs and CPs, focusing on both the underlying mechanisms of odour generation and the efficacy of state-of-the-art mitigation techniques. Malodours mitigation approaches including physical, chemical, and biological methods such as the addition of chemical agents, the use of microbial inoculants, the application of adsorbents and bulking agents and the modifications of operational parameters are explored and their performance critically evaluated. By integrating cost-effective odour control strategies into plant design and operational practices, WWTPs and composting facilities can achieve substantial reductions in odour emissions and compliance with stringent environmental regulations, while enhancing relationships with neighbouring communities. Finally, this review underscores the importance of a holistic and multi-disciplinary approach to odour management, combining scientific innovation with practical engineering solutions.

1. Introduction

The progressive increase in population, industrialization and consumerism are the main factors leading to the increase in waste generation in recent years. In this context, the volume of municipal solid waste and urban wastewater is increasing rapidly worldwide, which requires an integrated management to protect public health and the environment (Kim et al., 2019). Currently, odour pollution poses a significant threat to public health and the environment, which is particularly severe for residents of areas directly adjacent to industrial plants such as wastewater treatment plants (WWTPs) and composting plants (CPs) (Carrera-Chapela et al., 2014; Toledo et al., 2019a). This pollution has become one of the main environmental concerns in Europe, which has led to the development of a broad regulatory framework on air quality (European Commission, 2008). However, odour pollution regulations are widely diffused, with local entities facing social protests due to malodours nuisance (Caffyn, 2021; Wojnarowska et al., 2020). The recent construction of residential areas nearby the existing WWTPs and CPs is a source of problematic interactions, which needs to be evaluated from an odorous impact point of view. In this context, biological and physical-chemical treatments in WWTPs and CPs emit potentially toxic off-gases containing volatile organic compounds (VOCs), volatile sulphur compounds (VSCs) and volatile inorganic compounds (VICs), which negatively influence the local environment (He et al., 2018; Palmiotto et al., 2014; Schiavon et al., 2017). The formation of these unpleasant odours originates from microbial action during organic matter decomposition (e.g., food waste, sewage sludge, etc.) under anaerobic conditions (Talaiekhozani et al., 2016).

The perception of unpleasant odours remains subjective and can vary among individuals, necessitating the use of objective and precise measurement methods. For instance, Zhou et al. (2016) identified specific fecal odorants using Gas Chromatography-Mass Spectrometry (GC-MS), offering insights into the chemical composition of odour emissions. Similarly, Lewkowska et al. (2016) provided essential data on the

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* Corresponding author. Institute of Sustainable Processes, Dr. Mergelina s/n, 47011, Valladolid, Spain.
** Corresponding author. Institute of Sustainable Processes, Dr. Mergelina s/n, 47011, Valladolid, Spain.
E-mail addresses: z22topam@uco.es (M. Toledo), mutora@iq.uva.es (R. Muñoz).

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characteristics of odours emitted from wastewater treatment processes, employing field olfactometry as a practical tool for real-time odour evaluation. In this context, odour monitoring has gained significant importance since 2001, when odour impact was recognized in the second draft of the Biowaste Directive (European Commission, 2001). The European standard (EN 13725, 2003) established dynamic olfactometry as the reference method for determining odour concentration, expressed in European odour units per cubic meter (ou_E/m^3). This standard has since gained worldwide acceptance, solidifying dynamic olfactometry's role as a key tool in odour management. Recent research has highlighted dynamic olfactometry as a valuable technique for monitoring odour emissions from WWTPs and CPs (González et al., 2022). Dynamic olfactometry offers several advantages, including the reduction in the number of analyses required, minimizing time and cost, and providing a standardized approach for assessing odour emissions (Sironi et al., 2010). In fact, dynamic olfactometry, when combined with dispersion models, enables the quantification of odour impact, providing a comprehensive assessment of odour nuisance from industrial processes (Chin and Lindsay, 1994). Although not yet universally accepted as a substitute for dynamic olfactometry, electronic noses have also increasingly been highlighted in the literature as a promising tool for odour monitoring in WWTPs and CPs (Burgués et al., 2021; Sironi et al., 2007). The integration of electronic noses with dynamic olfactometry is often suggested to provide complementary benefits, enabling a more measurable and objective evaluation of odour nuisance (Littarru, 2007). These methodologies underline the importance of combining advanced analytical techniques with standardized monitoring approaches to address odour-related challenges effectively.

In WWTPs, the primary sources of malodorous emissions include primary and secondary clarifiers, aeration tanks, and sludge handling units (Senatore et al., 2021). The anaerobic digestion of sewage sludge is a massive source of methane (CH_4), volatile fatty acids (VFAs) and hydrogen sulphide (H_2S), the latter being highly odorous and toxic (Vu et al., 2022). Similarly, the composting process emits multiple odorous gases, including nitrogenous gases (mainly, NH_3 and N_2O) and VSCs, which arise from the microbial decomposition of organic waste (Sayara and Sánchez, 2021). Approximately 16%–94% of the initial total nitrogen contained in the composted waste is lost through these emissions (Awasthi et al., 2016). Additionally, about 50% of the total sulphur content is lost in the form of VSCs. Overall, the corrosive and toxic NH_3 and VSCs are dominant malodours during wastewater treatment and aerobic composting (Cui et al., 2022). Not all emitted gases contribute to odour nuisance, as some are odourless, such as CH_4 and CO_2. However, other gases, such as H_2S and NH_3, are highly odorous and can produce varying sensations depending on their concentrations and interactions with other substances. For instance, H_2S is detectable at very low concentrations, but its odour intensity decreases as it reacts with metals or oxygen to form non-volatile or less odorous compounds (Kashfi and Olson, 2012; Vakili et al., 2024). Similarly, NH_3 may interact with acidic compounds in the air, forming ammonium salts that lack odour (Renard et al., 2004). These reactions highlight the complexity of odour perception, where the chemical composition, reactivity, and environmental conditions collectively determine whether a gas is perceived as odorous or odourless. In addition, NH_3 is one of the primary compounds responsible for offensive odours and atmospheric pollution during the treatment of wastewater and composting of organic wastes containing a high nitrogen content (Pagans et al., 2006). While NH_3 threshold value suggests that it may not be immediately perceived at low concentrations, its concentrations in exhaust gases from wastewater and composting operations typically exceeds these thresholds.

A poor design and/or an inadequate operation rank among the most common causes of offensive odours in waste and wastewater management plants (Márquez et al., 2022; Toledo et al., 2018b). While some odour emissions can be mitigated through design and operational modifications, others are inherently difficult to control due to the nature of the treatment process unless a very complex odour control system is implemented. In this context, different odour prevention strategies have emerged and been implemented across wastewater and waste management processes. These strategies include the dosing of chemical agents (e.g., $FeCl_3$ and nitrate salts), the use of adsorbents and bulking agents (e.g., activated carbon, zeolite or biochar), the introduction of microbial inoculants (e.g., specific fungi or bacteria) and modifications in operational parameters (e.g., pH control and aeration).

The main objective of this review is to comprehensively examine and critically evaluate the strategies implemented to mitigate odour emissions during wastewater treatment and composting, with a particular focus on the nature of odorous compounds and the mechanisms of action involved in odour minimization. This review aims at elucidating the potential of dosing reagents, adsorbents and key microorganisms and modifying process operation to effectively address odour issues in both wastewater treatment and composting, while optimizing the overall treatment efficiency. Overall, this review seeks to provide a detailed assessment of state-of-the art strategies for odour management in order to identify effective practices that can be integrated into operational protocols to improve environmental quality and resource recovery during waste and wastewater management.

2. Odorants in wastewater treatment and composting plants

Odorous compounds emitted by domestic wastewater or by waste treatment processes can be both organic and inorganic in nature and are present as gases and vapours. The main inorganic odorant gases emitted from WWTPs and CPs are H_2S and NH_3 (Czarnota et al., 2023; Dinçer et al., 2020; Jiang et al., 2023). Although inorganic odorants and greenhouse gases are typically produced as a result of biological activity, VOCs and VSCs can also originate from the direct discharge of chemicals. Many of these odorants originate from the anaerobic decomposition of organic matter containing sulphur and nitrogen (Easter et al., 2005). These VOCs, VSCs and VICs are relatively small molecules with low molecular weights (typically below 200 g/mol) and possess different odour threshold values (OTVs), which entail that their scents are imperceptible below certain concentrations.

More specifically, indoles, skatoles, mercaptans, various VOCs, H_2S, and NH_3 are among the odorous compounds causing odour nuisance in wastewater collection systems and treatment facilities. The main odorous compounds in WWTPs and their corresponding odour, offensiveness and complaint threshold values (OFTV and OCTV, respectively) are listed in Table 1. The compounds included in the table were selected based on their relevance to WWTPs, as reported in the scientific literature. More specifically, compounds that are frequently identified in WWTP emissions were prioritized, and particularly those associated with the decomposition of organic matter and biological processes characteristic in these facilities. The selection was further guided by their significant contribution to odour emissions, considering their odour, offensiveness and complaint threshold values, which are critical for understanding their impact on odour nuisance. Additionally, compounds representing the major chemical groups commonly found in odorous emissions from WWTPs were included, such as VSCs (e.g., hydrogen sulphide, mercaptans), volatile nitrogen compounds (e.g., ammonia, amines), and other volatiles (e.g., organic acids). These odorants are typically characterized by a high degree of odour nuisance. From a qualitative point of view of odour nuisance, the frequency, intensity, duration, and hedonic tone of the selected compounds were described using findings from relevant studies (Czarnota et al., 2023). Thus, methylamine and butanone are the only compounds with a low odour nuisance. Acetone, ammonia, pyridine, propionic acid, and acetic acid exhibit a moderate odour annoyance. Moreover, many of these listed chemicals generate odours similar to rotting vegetables, while others emit smells resembling fecal matter, sweat, or rotten eggs. It is important to highlight that most sulphur-containing compounds, nitrogen-containing compounds (i.e., trimethylamine), volatile fatty acids and other organic compounds (i.e., indole and skatole) exhibit the lowest OTV, OFTV and OCTV and therefore, very low concentrations of these odorants can be easily detected by the human olfactory system.

In CPs, the type of VOCs and their emissions flowrates vary significantly throughout the composting process. A diverse range of VOCs and odours are typically generated during composting, most of them emerging during the early stages of the process (i.e., thermophilic phase) (Toledo et al., 2018a). The highest concentrations of total volatile organic compounds (TVOCs) are recorded in the early stages of composting, with levels reaching 3000–14,000 mg/m^3 (1.5–2 times greater than those in the middle and late stages) (Kumar et al., 2011; Turan et al., 2007). Many VOCs are malodorous, and some can be toxic, irritating, and carcinogenic, posing a serious threat to human health (Nicell, 2009), and causing air pollution that further impacts on natural ecosystems global climate change. The wide diversity of compostable materials entails the production of different VOCs, including methyl sulphide, dimethyl disulphide, methanethiol, ethylbenzene, methylamine, ammonia, acetic acid, and dimethyl trisulphide, among others (Ki et al., 2018; Mao et al., 2006; Zhu et al., 2016). Table 1 summarizes the most common odorants during composting process and their corresponding OTV, OFTV, OCTV, and degree of nuisance, among others.

Malodorous emissions from WWTPs are influenced by factors such as atmospheric pressure, air turbulence above the source, treatment plant size and configuration, and the flowrate, composition, pH, and temperature of the incoming wastewater, and the operational practices of the plant (including dissolved oxygen concentration) (Byliński et al., 2019; Wang et al., 2012). Interestingly, odour emissions from WWTPs exhibit seasonal variations, with ambient temperature and organic content in wastewater playing a significant role on the profile and concentration of odorants (Ruiz-Muñoz et al., 2023). Thus, increased temperatures during the summer months significantly boost odour emissions (Asadi and McPhedran, 2021), which are inherently linked to the sewage transport and treatment processes, including sewer networks, rising mains, primary settlers, and sludge processing. Odorants are primarily formed by microbial decomposition of organic matter, especially under anaerobic conditions, which results in the highest levels of odorant emissions (Lebrero et al., 2011).

The influence of pH on odour emissions from WWTPs is also significant since it governs the dissociations of H_2S, NH_3 and VFAs. Lower pH levels in sewage can increase the volatility and concentration of dissolved H_2S and non-dissociated VFA, leading to higher H_2S emissions. In this context, Yan et al. (2018) showed that a decrease in the pH of sewage increased the amount of H_2S released from wastewater. Similarly, Chen et al. (2018) found that H_2S concentrations in the WWTP headworks were negatively correlated with sewage pH. When the sewage pH exceeded 7.2, H_2S concentrations decreased below 5 mg/m^3. On the other hand, high pH levels promote the emission of higher concentrations of NH_3 (Blanes-Vidal et al., 2012). Therefore, the management of pH in the different operational units of a WWTP is crucial for controlling odour emissions and minimizing environmental and public health impacts.

Odour emissions in composting facilities are influenced by factors such as moisture content, aeration, temperature, carbon-to-nitrogen (C:N) ratio, pH levels, feedstock composition, particle size, microbial activity, and turning frequency (Meena et al., 2021). In this sense, the moisture content of the compost influences microbial metabolism and physical structure of solid matrices during composting. Maintaining moisture content between 50 and 60% and ensuring adequate aeration promotes aerobic conditions, which are essential to support an effective microbial activity and decomposition of organic matter (Kim et al., 2016). In contrast, a high moisture content typically induces anaerobic conditions, causing the production of odorous compounds like NH_3 and H_2S. On the other hand, despite high temperatures during the thermophilic phase (50–70 °C) are beneficial for pathogenic bacteria inactivation (Wichuk and McCartney, 2007), the increased biological activity favours odour formation and release. Indeed, the rapid breakdown of easily biodegradable organic matter along with nitrogen and sulphur-based compounds during the thermophilic phase of the composting process results in the formation and release of larger mass flowrates of odorous compounds (González et al., 2020). Finally, optimal C:N ratios and particle sizes support efficient aerobic microbial activity, while balanced pH levels (7–8.5) prevent the production and emissions of odorous compounds. Conversely, extreme pH values, poor aeration, and irregular pile turning can create anaerobic zones, leading to the generation of odorants. Therefore, an appropriate control of these operational factors is crucial to minimize odorant formation and emissions during composting.

3. Mechanisms of odorant generation in wastewater treatment and composting plants

Odour pollution in WWTPs and CPs is mainly associated with the release of VSCs, VICs and VOCs, which are typically generated as a result of the anaerobic decomposition of organic matter by microbial communities. Understanding the mechanisms of VSC, VIC and VOC formation and emissions is crucial for developing effective strategies to control and reduce the release of harmful odorants during wastewater treatment and composting.

Multiple studies have investigated VSC emissions from WWTPs, focusing on their detection in both ambient air and wastewater (Barczak et al., 2022; Guerrero et al., 2025). The concentrations of VSCs in ambient air exhibit a significant variability as a function of the sampling location and atmospheric conditions. A comprehensive analysis of 24 studies monitoring VSC concentrations from different WWTP operational units, including grit chambers, screens, primary settlers, and biological reactors, revealed a wide range of compounds and concentrations. More specifically, H_2S in WWTPs typically ranges from 0.1 to 20480 μg m^{-3}, methyl mercaptan (MM) from 0.4 to 2.4 μg m^{-3}, dimethyl sulphide (DMS) from 0.4 to 5450 μg m^{-3}, carbon disulphide (CS_2) from 3.06 to 9.82 μg m^{-3}, and dimethyl disulphide (DMDS) from 0.62 to 1600 μg m^{-3} (Jiang et al., 2017). Similarly, reference concentrations of dissolved VSCs in domestic wastewaters have been estimated at approximately 255 μg L^{-1} for MM, 28.48 μg L^{-1} for DMS, 3.21 μg L^{-1} for CS_2, and 102.81 μg L^{-1} for DMDS (Lee and Brimblecombe, 2016). These estimates are based on monitoring results from influent, primary effluent, and secondary effluent in WWTPs. However, the accurate determination of VSC concentrations in wastewater is challenging due to their rapid degradation and transformation during the wastewater treatment processes. Temperature significantly influences VSC emission levels, impacting on microbial metabolism, dissolved oxygen levels, and VSC solubility in wastewater (Chen and Szostak, 2013). High temperatures trigger microbial metabolism, resulting in high VSC production. Additionally, elevated temperatures reduce dissolved oxygen solubility, leading to an increased VSC generation if oxygen levels are insufficient. In addition, higher VSC concentrations are released into the air as their aqueous solubility decreases at higher temperatures.

H_2S and MM are the main VSCs produced during wastewater treatment, which are further released and transformed through air exposure and mechanical disturbances (Hvitved-Jacobsen et al., 2013). In fact, methylation of MM is the dominant mechanism for endogenous DMS production (Kiene and Hines, 1995), while DMDS is chemically synthesized through the abiotic oxidation of MM (Chin and Lindsay, 1994) according to equations (1) and (2), respectively.

$$R - O - CH_3 + CH_3SH \rightarrow R - OH + CH_3SCH_3\ (DMS) \tag{1}$$

$$2CH_3SH + 0.5O_2 \rightarrow H_2O + CH_3S_2CH_3\ (DMDS) \tag{2}$$

According to Fig. 1, the sulphide cycle during wastewater treatment involves the interplay between sulphide-oxidizing bacteria (SOB) and sulphate-reducing bacteria (SRB) across both aqueous and gaseous phases (Toledo and Muñoz, 2022a). In the anaerobic zones of the wastewater, SRB reduce sulphate (SO_4^{2-}) to H_2S using organic matter or H_2 as electron donor, which dissolves in the wastewater. Due to its high

Table 1
Odour, offensiveness and complaint threshold values and main characteristics of common odorous compounds in WWTPs and CPs.

Odorous compounds	Substance	Facility (WWTP/CP)	Chemical formula	Molecular weight (g/mol)	Odour characteristic	OTV[a] ($\mathrm{ppm_v}$)	OFTV[b] ($\mathrm{ppm_v}$)	OCTV[b] ($\mathrm{ppm_v}$)	Concentration with potential impact on humans LTEL/STEL (mg/m^3)	Degree of nuisance[d]
Sulphur containing compounds	Hydrogen sulphide	WWTP/CP	H_2S	34.10	rotten eggs	0.00041	0.00009-0.0001	0.00009-0.0001	7/14	High
	Methyl mercaptan (Methanethiol)	WWTP	CH_4S	48.10	stinky, rotten cabbage, radish	0.000070	0.00018	0.00018	1/2	High
	Ethyl mercaptan (Ethanethiol)	WWTP	C_2H_6S	62.13	stinky, skunk secretion, garlic	0.0000087	0.0000079-0.0000097	0.0000079-0.0000097	1/2	High
	Benzyl mercaptan	WWTP	$C_6H_5CH_2SH$	124.21	stinky, strong, disgusting	0.00019	0.00027-0.00036	0.00040-0.00054	n.d.	High
	Butyl mercaptan (Butano-1-thiol)	WWTP	$C_4H_{10}S$	90.18	rotten cabbage, skunk secretion, mustard	0.0000028	n.d.	n.d.	1/2	High
	Carbon disulphide	CP	CS_2	76.14	rotten eggs	0.026-0.038[b]	0.029-0.041	0.033-0.046	15/n.d.	High
	Methyl propyl disulfide	CP	$C_4H_{10}S$	122.25	cooked onion/garlic	n.d.	n.d.	n.d.	n.d.	n.d.
	Diethyl sulphide	WWTP	$C_4H_{10}S$	90.19	rotten vegetables, garlic, stinky	0.000033	n.d.	n.d.	n.d.	High
	Dimethyl trisulphide	CP	$C_2H_6S_3$	126.30	mature cheese-like, onion, garlic	n.d.	n.d.	n.d.	n.d.	n.d.
	Allyl methyl sulphide	CP	C_4H_8S	88.17	garlic	0.00022[c]	n.d.	n.d.	n.d.	n.d.
	Allyl mercaptan (2-Propentiol)	WWTP/CP	C_3H_6S	47.15	stink, garlic, coffee	0.2[c]	n.d.	n.d.	n.d.	High
	Methyl propenyl sulphide	CP	C_4H_8S	88.17	acidic garlic	n.d.	n.d.	n.d.	n.d.	n.d.
	Dimethyl sulphide	WWTP	C_2H_6S	62.13	rotten vegetables, cabbage, turnips, stinky	0.0030	0.0032	0.0036	n.d.	High
	Dimethyl disulphide	WWTP/CP	$C_2H_6S_2$	94.20	stinky, disgusting	0.0022	0.0022	0.0026	2.5/5	High
Nitrogen containing compounds	Ammonia	WWTP/CP	NH_3	18.01	suffocating, pungent, irritating, ammoniacal	1.5	0.9	1.1	14/28	Medium
	Methylamine	WWTP	CH_3NH_2	31.10	ammoniacal, fishy	0.035	n.d.	n.d.	5/15	High
	Dimethylamine	WWTP	$(CH_3)_2NH$	45.08	ammoniacal, fishy	0.033	0.097-0.126	0.115-0.154	3/9	High
	Trimethylamine	WWTP/CP	C_3H_9N	59.11	ammoniacal, fishy	0.000032	0.00013	0.00013	4.9/12.5	High
	Pyridine	WWTP	C_5H_5N	79.10	faint, sweetish, unpleasant	0.063	0.063-0.077	0.063-0.077	14/5	Medium
	2-methylpyridine	CP	C_6H_7N	93.13	faint, sweetish, unpleasant	n.d.	n.d.	n.d.	n.d.	Medium
Terpenes	Limonene	CP	$C_{10}H_{16}$	136.24	pleasant orange	0.038	n.d.	n.d.	25/50	Medium
	Camphor	CP	$C_{10}H_{16}O$	152.23	strong mothball-like odour	0.079	n.d.	n.d.	12/19	High
Ketones	Acetone	WWTP/CP	C_3H_6O	58.08	sweet, musty, fruity, ethereal	42	5.25-7.50	7.09-8.37	600/1800	Medium
	Butanone	WWTP/CP	C_4H_8O	72.11	pungent, minty, similar to acetone	0.44	n.d.	n.d.	n.d.	Low
	4,4-dimethyl-2-pentanone	CP	$C_7H_{14}O$	114.19	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
	3-ethylcyclopentanone	CP	$C_7H_{12}O$	112.17	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
	3-methyl-2-hexanone	CP	$C_7H_{14}O$	114.19	peppermint-like	n.d.	n.d.	n.d.	n.d.	n.d.
Furans	2,4-dimethylfuran	CP	C_6H_8O	96.13	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
	2-pentylfuran	CP	$C_9H_{14}O$	138.21	beany, grassy odour	1	n.d.	n.d.	n.d.	n.d.
Alkanes	butylcyclohexane	CP	$C_{10}H_{20}$	140.27	floral woody	n.d.	n.d.	n.d.	n.d.	Low
	heptane	CP	C_7H_{16}	100.21	gasoline	0.67	5.7-6.9	8.6–10.5	400/500	Low
	decane	CP	$C_{10}H_{22}$	142.29	gasoline	0.62	n.d.	n.d.	n.d.	n.d.
	2,6-dimethyldecane	CP	$C_{12}H_{26}$	170.33	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
	undecane	CP	$C_{11}H_{24}$	156.31	gasoline	0.87	n.d.	n.d.	n.d.	Medium

(*continued on next page*)

Table 1 (*continued*)

Odorous compounds	Substance	Facility (WWTP/CP)	Chemical formula	Molecular weight (g/mol)	Odour characteristic	OTV[a] (ppm_v)	OFTV[b] (ppm_v)	OCTV[b] (ppm_v)	Concentration with potential impact on humans LTEL/STEL (mg/m^3)	Degree of nuisance[d]
	dodecane	CP	$C_{12}H_{26}$	170.34	gasoline	0.11	n.d.	n.d.	n.d.	Medium
	butane	CP	C_4H_{10}	58.12	faint petroleum	1200	n.d.	n.d.	750/600	n.d.
	isobutane	CP	C_4H_{10}	58.12	faint petroleum	n.d.	n.d.	n.d.	1000/800	n.d.
	pentane	CP	C_5H_{12}	72.15	gasoline	1.4	n.d.	n.d.	600/1000	n.d.
Aromatic compounds	p-xylene	CP	C_8H_{10}	106.16	sweet	0.058	n.d.	n.d.	50/100	n.d.
	o-xylene	CP	C_8H_{10}	106.16	sweet	0.38	n.d.	n.d.	50/100	n.d.
	styrene	CP	C_8H_8	104.15	sweet	0.035	0.044-0.063	0.044-0.063	100/n.d.	n.d.
	benzene	CP	C_6H_6	78.11	sweet	2.7	2.7-3.62	2.7-3.62	5/0.5	n.d.
Halogenated compounds	1-chloro-2-propanol	CP	C_3H_7ClO	94.54	ether	n.d.	n.d.	n.d.	n.d.	n.d.
	tetrachloroethylene	CP	C_2Cl_4	165.83	ether, sweet	1	n.d.	n.d.	40/20	n.d.
Esters	methyl acetate	CP	$C_3H_6O_2$	74.08	fruity	1.7	n.d.	n.d.	250/200	n.d.
	methyl propionate	CP	$C_4H_8O_2$	88.11	fruity, sweet	0.098	n.d.	n.d.	n.d.	n.d.
Alcohols	2-butoxyethanol	CP	$C_6H_{14}O_2$	118.17	ether, sweet	0.043	n.d.	n.d.	50/20	Low
	2-butanol	CP	$C_4H_{10}O$	74.12	fruity	0.038	n.d.	n.d.	150/n.d.	n.d.
	1-butoxy-2-propanol	CP	$C_7H_{16}O_2$	132.20	ether	0.16	n.d.	n.d.	40/20	n.d.
Volatile fatty acids	Acetic acid	WWTP	$C_2H_4O_2$	60.05	pungent, acetic	0.0060	0.66-0.80	0.78-1.00	25/50	Medium
	Propionic acid	WWTP	$C_3H_6O_2$	74.08	pungent, rancid, irritating	0.0057	0.0134-0.0150	0.0202-0.0227	30/45	Medium
	Butyric acid	WWTP	$C_4H_8O_2$	88.11	rancid butter, sweat	0.00019	0.005-0.006	0.005-0.006	n.d.	High
Other organic compounds	Indole	WWTP	C_8H_7N	117.15	rotten proteins, faeces, faeces	0.00030	n.d.	n.d.	n.d.	High
	Skatole	WWTP	C_9H_9N	131.17	faeces, faeces	0.0000056	n.d.	n.d.	n.d.	High

OTV: Odour Threshold Value; OFTV: Offensiveness Threshold Value; OCTV: Complaint Threshold Value; LTEL: Long-Term Exposure Limit; STEL: Short-Term Exposure Limit; n.d.: no data.

[a] (Nagata, 2003).

[b] (Bokowa, 2022).

[c] (Ruth, 1986).

[d] (Czarnota et al., 2023).

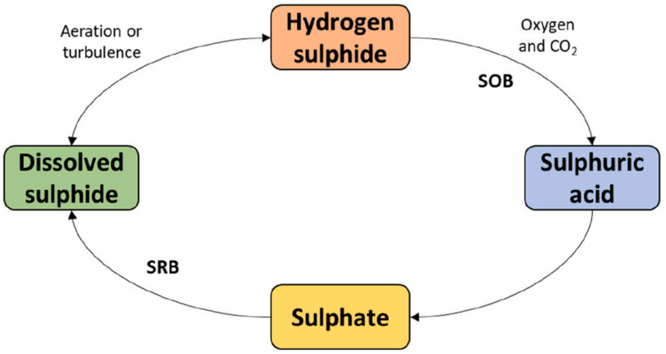

Fig. 1. Sulphide cycle in the gaseous and aqueous phases in wastewater (Toledo and Muñoz, 2022a,b).
SOB: sulphur oxidizing bacteria; SRB: sulphur reducing bacteria.

volatility, H_2S rapidly partitions into the headspace in equilibrium with the wastewater (Eq. (3)). Under aerobic conditions, SOBs, such as *Thiobacillus* and *Beggiatoa*, oxidize H_2S to elemental sulphur (S^0) (Eq. (4)) or SO_4^{2-} (Eq. (5)), completing the cycle. This oxidation can occur both in the aqueous phase and at the gas-liquid interface. Environmental conditions such as pH, temperature, and oxygen availability govern the sulphur cycle (Moloantoa et al., 2023). Indeed, a low pH favours the formation of volatile H_2S, while higher temperatures enhance microbial activity and gaseous H_2S volatilization. Thus, the sulphide cycle involves a dynamic set of equations governed by environmental conditions, microbial processes, and the physical-chemical characteristics of the wastewater.

$$SO_4^{2-} + organic\ matter \xrightarrow{SRB} H_2S + CO_2 + Other\ products \tag{3}$$

$$2H_2S + O_2 \xrightarrow{SOB} 2S^0 + 2H_2O \tag{4}$$

$$H_2S + 2O_2 \xrightarrow{SOB} SO_4^{2-} + 2H^+ \tag{5}$$

According to the Environmental Protection Agency (EPA) of the United States, VOCs are chemical compounds characterized by a high vapor pressure and low water solubility (EPA, 2022). These properties allow VOCs to easily strip-out from solid and liquid phases into the atmosphere. VOC emissions from WWTPs may occur through a variety of mechanisms which mainly include volatilization, sorption, chemical reactions and biodegradation (EPA, 1994). More specifically, volatilization occurs when dissolved VOCs transfer from the aqueous phase to the gaseous phase, facilitated by high temperatures and aeration rates. Sorption involves VOCs attachment onto solid particles, influenced by their hydrophobicity. Chemical reactions such as oxidation, hydrolysis, and photolysis also play a role in VOC emissions from WWTP. For instance, oxidants like ozone and chlorine can react with VOCs, leading to oxidized byproducts. Hydrolysis can break down VOCs in the presence of water, while photolysis can degrade them through sunlight exposure. Finally, biodegradation is based on microorganisms capable of metabolizing VOCs aerobically or anaerobically. Aerobic biodegradation relies on oxygen as electron acceptor to transform VOCs into carbon dioxide (CO_2), water (H_2O), and biomass, whereas anaerobic biodegradation produces CH_4 and intermediate compounds like VFAs.

Fig. 2 depicts the metabolic pathways of VOCs and VSCs during composting. For instance, alkanes are synthesized during the initial stage of composting and the decomposition of organic matter (i.e., mainly from cellulosic materials), and then released at high internal temperatures and humidity (Tan et al., 2017). The high temperature and microbial activity during the early stages of composting supports the production of organic compounds containing chlorine (Cl) and bromine (Br), and a large number of hydrogen atoms on alkanes can be replaced by Cl and Br, generating a variety of halogenated compounds through substitution reactions, which subsequently produce alcohols by hydrolysis reactions (Mustafa et al., 2017). Moreover, alcohols can be chemically oxidized to produce ketones under aerobic conditions. In this context, oxygen is a key parameter governing the degradation of organic matter and its concentration influences the composition of VOCs during composting process (Zheng et al., 2021). The oxygen consumption in the early stage of composting is high, leading to a semi-consumptive and semi-anaerobic state in the composting piles, which ultimately induce the formation of large amounts of H_2S. In the metabolic route of H_2S formation, MM can be generated by combination with alcohols, and/or can also be oxidized to generate sulphur-containing compounds such as DMDS (Eq. (2)). In the anaerobic zone of composting piles, large amounts of DMS, DMDS, H_2S, carbonyl sulphide (COS), CS_2 and MM are synthesized by SRB and from the decomposition of sulphur-containing organic matter under anaerobic conditions (Duan et al., 2022). On the contrary, in the aerobic zone of the composting piles, aerobic microorganisms are responsible of the synthesis of high emissions of alkanes and halogenated compounds, such as n-tridecane, n-tetradecane, 1,3-dichlorobenzene, bromobenzene, 2-chlorotoluene, and of sulphur-containing compounds to a lower extent. In the initial stages of the composting process, halogenated hydrocarbons can generate alkenes at high temperatures, which can form aromatic compounds by addition reaction (Atkinson and Arey, 2003). However, the conversion of alkenes to aromatic compounds by addition reactions is more demanding (H. Chen et al., 2018). In the middle and late stages of composting, oxygen

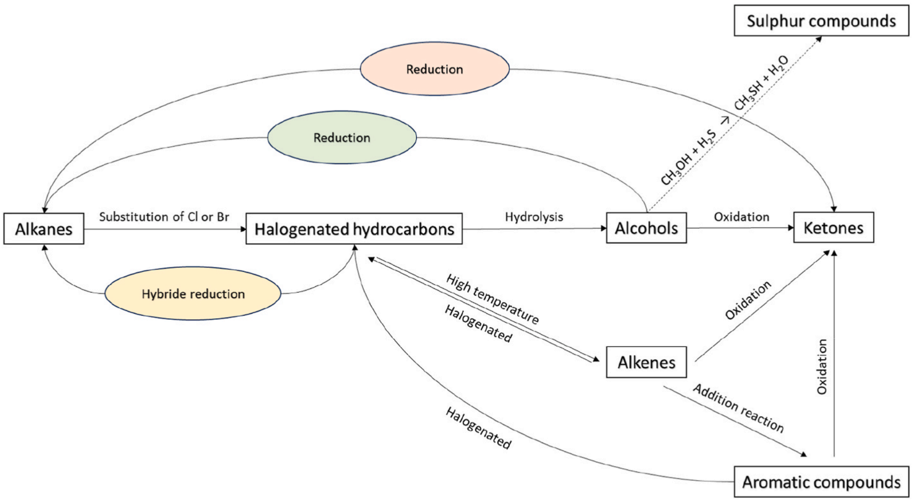

Fig. 2. Metabolic pathways of volatile organic compounds (VOCs) and volatile sulphur compounds (VSCs) during the composting process.

consumption gradually decreases, leading to an increase in the concentration of oxygenated compounds emitted. As the temperature of the pile gradually decreases, the concentration of alkanes produced from cellulose degradation decreases and the total emissions of VOCs drops.

4. Odour prevention strategies in wastewater treatment

WWTPs play a crucial role in safeguarding public health and the environment by treating wastewaters prior to discharge. However, these facilities frequently encounter challenges in preventing and managing odour nuisance, which can adversely affect the nearby populations and ecosystems. Odour prevention strategies in WWTPs involve a combination of chemical and biological methods to minimize the formation and release of unpleasant odours. Table 2 summarizes the main odour prevention strategies in WWTPs as well as the reaction mechanisms involved, dosage applied and the odour removal potential.

Chemical and biological dosing are recognized as effective odour prevention strategies to cope with odour, corrosion, and greenhouse gas (GHG) emissions problems in sewer systems and WWTPs (Ganigue et al., 2011; Toledo and Muñoz, 2022a,b). This approach primarily involves the addition of chemicals to the liquid phase of the wastewater to inhibit the formation and release of odorous compounds. Conventional strategies include air/oxygen injection, and the addition of nitrate salts (e.g., $NaNO_3$, $Ca(NO_3)_2$ and $Fe(NO_3)_3$), iron salts (e.g., $FeSO_4$, $FeCl_2$, $FeCl_3$, $Fe(NO_3)_3$ and $Fe_2(SO_4)_3$), alkali (e.g., $Mg(OH)_2$ and NaOH), hydrogen peroxide (H_2O_2), chlorine, and potassium permanganate ($KMnO_4$). This section discusses the underlying mechanisms, effectiveness and potential limitations of these odour control strategies. Table 3 summarizes the key features, advantages, and disadvantages of each odour prevention strategy discussed in this section.

4.1. Injection of air or pure oxygen

The injection of air or pure oxygen into the wastewater is a well known method for preventing odours in WWTPs (Jordan et al., 2012). This technique works by promoting aerobic conditions within the wastewater, thereby inhibiting the formation of odorous compounds that typically arise under anaerobic conditions (i.e., mainly H_2S and VFAs). In fact, improper aeration can disrupt microbial activity, leading to incomplete stabilization of organic matter in sludge. This imbalance not only degrades sludge quality but also promotes the release of additional odorants, including NH_3 and H_2S to a greater extent, as well as indole, pinene, p-cresol, and trimethylamine to a lesser extent, due to anaerobic conditions or excessive volatilization (Fisher et al., 2019).

Aeration can be achieved through different methods, including diffused aeration systems, surface aerators, and the direct injection of pure oxygen in pre-treatment and primary treatment units. More specifically, oxygen supports the growth of aerobic microorganisms that outcompete anaerobic bacteria, thus reducing the production of odorous compounds such as VICs, VSCs and VFAs. In the case of air injection, Ochi et al. (1998) reported that air injection completely eliminated dissolved sulphide at the outlet of the sewer network when dissolved oxygen (DO) in the wastewater was above 0.2 mg O_2/L. Other authors have reported that the occurrence of sulphide in sewage can be avoided only when DO is above 0.5 mg O_2/L (Talaiekhozani et al., 2016). Due to the limited oxygen solubility (~8 mg/L in equilibrium with air and ~40 mg/L in equilibrium with pure O_2), this strategy is mainly implemented in rising mains rather than in gravity sewers, and pressurized air is typically required to increase DO concentration in sewage (Zhang et al., 2008).

Alternatively, pure oxygen injection is used, which would deliver oxygen concentrations five times higher than those achieved with air (Zhang et al., 2008). The injection of pure oxygen into the wastewater is commonly employed in certain aeration systems to improve oxygen transfer efficiency, particularly when the objective is to reach higher DO levels during high-strength wastewater treatment. When air is added to wastewater, it can strip H_2S and/or VOCs that might be dissolved. Compared to the traditional compressed air approach, pure oxygen injection can result in improved mass transfer since it is typically transferred at a high rate (70–90%) and there is no nitrogen to interfere. These factors help to minimize the potential stripping that is typical of air-based systems where VOCs are present. Recent studies reported the effectiveness of direct injection of pure oxygen into the sewer system, with a reduction in soluble sulphides from 35 mg/L down to 2 mg/L and

Table 2
Overview of odour prevention strategies in wastewater treatment plants: Reaction mechanism, dosage or ratio applied and odour reduction capacity.

Odour prevention strategy	Reaction mechanism	Dosage or ratio applied	Odour reduction capacity	References
Air or pure oxygen	Promotes aerobic conditions inhibiting the formation of odorous compounds	Aeration rates at 0.2 mg O_2/L or higher Pure oxygen is five times more soluble than air in water, reaching up to 40–50 mg O_2/L	Reduction of influent soluble sulphides from 35 mg/L to 2 mg/L	Zhang et al. (2008)
Nitrate salts	Microbiological oxidation of odorants by SOB using nitrate as electron donor	Addition of nitrate at concentrations of 10–40 mg/L	Reduction of H_2S and acetic acid by 95% and 42%, respectively	(Toledo and Muñoz, 2023; Yang et al., 2005)
Iron salts	Chemical oxidation of sulphide to S^0	Molar ratio of Fe:$S = 1.3$:1 for ferrous salts, Fe:$S = 0.9$:1 for ferric salts or a mixture of both salts at a ratio of Fe^{3+}:$Fe^{2+} = 2$:1	Reduction of sulphide concentration lower than 0.1 mg S/L at pH around 7.0	Firer et al. (2008)
Alkali	Increases the pH in wastewater, reducing the volatilization of H_2S from the liquid phase to the vapor phase Inhibits SRB activity	Increase the pH to the range up to 9.0–10.5	$Mg(OH)_2$ dosing increases the sewage pH to 8.8 ± 0.1, reducing sulphide concentration to 6.7 ± 0.9 mg S/L. CH_4 emissions are also reduced by 58.0% ± 4.9%.	Cen et al. (2023)
Hydrogen peroxide	Chemical oxidation of sulphides and sulphites to into SO_4^{2-}	1 and 2 mg H_2O_2 are required per 1 mg of sulphide	Odour concentration of raw wastewater (2980 ± 110 ou_E/m^3) can be reduced by 96.3% ± 1.9% to a level of 100 ± 15 ou_E/m^3	Dębowski et al. (2022)
Chlorine	Chemical oxidation of H_2S and other odorous compounds Reduces microbial populations that can contribute to odour formation by disinfection	Addition of 4.2 g chlorine per gram of sulphide	Total sulphide removal	Steiner et al. (2010)
Potassium permanganate	Powerful and versatile oxidant that can be used to destroy VSCs, phenols, among others	Typical dosage range for $KMnO_4$ is 6–7 parts per part of sulphide pH for $KMnO_4$ addition is typically maintained in the range of 8.0–9.5	Reduction of H_2S levels from 815 to 0 ppm	Wlodarchak et al. (2012)

Table 3
Key features, advantages, and disadvantages of each odour prevention strategy in wastewater treatment plants.

Treatment	Key Features	Advantages	Disadvantages	Reference
Injection of air or pure oxygen	Introduction of air or pure oxygen to promote aerobic conditions in wastewater	Prevents formation of odorous compounds (e.g., H_2S, VFAs) Supports aerobic bacteria, reducing anaerobic bacteria	Limited oxygen solubility (~8 mg/L in air, ~40 mg/L in pure oxygen) Requires pressurized air for effectiveness	Zhang et al. (2008)
Addition of nitrate salts	Nitrate salts facilitate microbial oxidation of sulphides to less odorous compounds	More effective than air injection for sulphide control. Higher aqueous solubility than oxygen. Reduces H_2S and acetic acid emissions	Requires precise dosing Can have competing reactions (e.g., ammonia in the wastewater)	Yang et al. (2005)
Addition of iron salts	Iron salts (e.g., ferric chloride) oxidize sulphide to elemental sulphur or sulfates.	High effectiveness in sulphide control Inhibits sulphate-reducing and methanogenic activities	pH-sensitive, requiring pH control High operational costs due to iron salt excess.	Firer et al. (2008)
Addition of alkali	Addition of alkali (e.g., $Mg(OH)_2$, NaOH) to increase pH and reduce H_2S volatilization	Inexpensive and simple to apply $Mg(OH)_2$ provides sustained odour control Reduces SRB activity by 25–50%	Overdosing can cause excess unreacted alkalinity Intermittent NaOH dosing may require further optimization	(Cen et al., 2023; Gutierrez et al., 2009)
Addition of hydrogen peroxide	H_2O_2 is used as a strong oxidizing agent to reduce H_2S and other odorous compounds	Rapid reaction time Effective in reducing BOD, COD, and offensive odours No harmful by-products	Hazardous chemical, requiring special handling and containment Needs careful dosing to avoid overuse	Ksibi (2006)
Addition of chlorine	Chlorine is used for both disinfection and odour control by oxidizing sulphides	Relatively inexpensive and easy to apply Effective against a wide range of pathogens	Health and safety concerns due to toxic by-products (e.g., trihalomethanes, haloacetic acids) Corrosive	Freitas et al. (2021)
Addition of potassium permanganate	Potassium permanganate is used as a strong oxidant to eliminate sulphides and other compounds.	Broad pH range (acidic, neutral, alkaline) for oxidation. Reduces H_2S and organic compounds effectively	By-products include manganese dioxide sludge, which can cause environmental and operational issues	Medialdea et al. (2005)

the concomitant reduction of biological oxygen demand (BOD) concentrations from 240 mg/L down to 208 mg/L (Cipriani et al., 2018). Thus, dosing oxygen in pumping stations of sewer networks could achieve better sulphide removal than air injection (Gutierrez et al., 2008).

Existing studies primarily demonstrate the efficacy of air or pure oxygen injection in sulphide control but fall short in optimizing cost-effective, large-scale applications for diverse wastewater systems. Future research should focus on integrating advanced aeration technologies, such as nanobubble systems or membrane aeration, to enhance oxygen transfer efficiency and reduce energy consumption.

4.2. *Addition of nitrate salts*

Nitrate salts are widely utilized for sulphide control in sewers and WWTPs (Auguet et al., 2015; Toledo and Muñoz, 2023). This odour prevention strategy facilitates the microbial oxidation of sulphide to S^0 or SO_4^{2-} in the presence of nitrate mediated by SOB, as described in Eq. (6) and Eq. (7), or prevents the occurrence of septic conditions. Given its higher aqueous solubility compared to oxygen, when nitrate is dosed appropriately into the wastewater it can provide more effective sulphide control than oxygen injection. Despite the low aqueous solubility of S^0 (approximately 5 μg/L at 25 °C), elemental sulphur can react with biogenic sulphide to form polysulphides. This reaction enhances the bioavailability of S^0, thereby accelerating sulphur reduction processes (Zhang et al., 2021).

$$5S^{2-} + 2NO_3^- + 12H^+ \rightarrow 5S^0 + N_2 + 6H_2O \tag{6}$$

$$5S^0 + 6NO_3^- + 2H_2O \rightarrow 5SO_4^{2-} + 3N_2 + 4H^+ \tag{7}$$

Previous studies demonstrated that dosing nitrate salt at concentrations of 10–40 mg N/L efficiently reduced sulphide concentrations in wastewaters to 0.2–3.0 mg S/L (Yang et al., 2005). Similarly, effluents rich in nitrates or nitrites (e.g., the nitrified wastewater from the nitrification tank or effluents with high ammonia concentration like centrates from the anaerobic digestion of mixed sludge) could be recycled at the headworks of the WWTP in order to provide additional electron acceptors for microorganisms to oxidize the dissolved odorants (Kiene and Hines, 1995). In this context, Toledo and Muñoz (2023) combined oxidized nitrogen recycling (ONR) with activated sludge recycling (ASR) (i.e., activated sludge recycled from the secondary settler or mixed liquor of the nitrification tank into the inlet of the pilot WWTP headworks) to effectively prevent the emission of odorous compounds in the primary settler, achieving reductions of 95 % and 42 % for H_2S and acetic acid, respectively. Fig. 3 shows the schematic diagram to implement ONR from centrate nitrification units (ONR (1)) and from the nitrification stage in the biological reactor (ONR (2)), and the implementation of ASR from the secondary settler (ASR (1)) and from the aerobic activated sludge reactor (ASR (2)). The combined implementation of ASR and ONR was shown the most promising strategy, favouring the microbiological oxidation of H_2S and acetic acid by SOB and heterotrophic bacteria using the recirculated nitrate or nitrite as electron donor, being the main mechanism governing the minimization of both odorants.

While nitrate salts effectively control sulphide concentrations, the long-term implications of nitrate dosing on microbial communities and the potential formation of harmful intermediates like nitrite have not been sufficiently studied. Further investigation into hybrid approaches that combine nitrate dosing with other treatments, such as bioaugmentation, could provide more sustainable solutions.

4.3. *Addition of iron salts*

Iron salts are commonly used both for phosphorus removal and as coagulants for the removal of suspended solids (Metcalf, 1991). Different iron salts have been utilized to control sulphide in sewers and WWTPs including ferrous chloride, ferric chloride and in some cases ferrous sulphate (Bertran De Lis et al., 2007; Ganigue et al., 2011; Jameel, 1989; Padival et al., 1995). Previous studies have validated the use of ferric salts in sewer systems for sulphide and phosphate removal from wastewater (Gutierrez et al., 2010). In addition, the effectiveness of ferric chloride dosage on sewage sludge in the control of malodourous sulphur gas emissions was also reported by Devai and Delaune (2002).

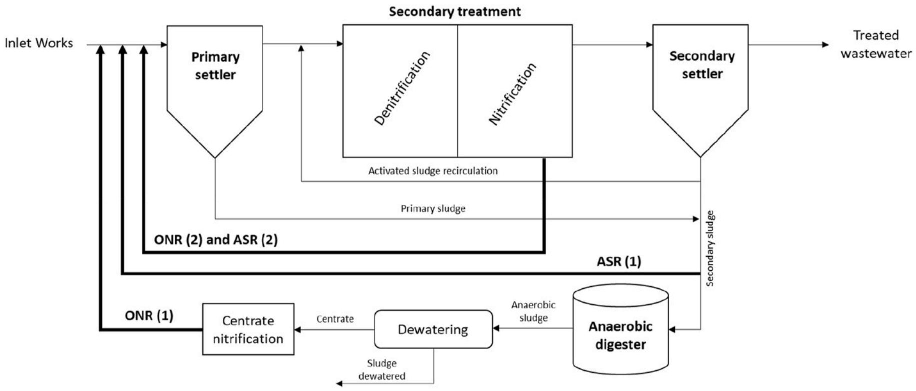

Fig. 3. Flow diagram of activated sludge recycling (ASR) and oxidized nitrogen recycling (ONR).

However, to the best of our knowledge there is no reported study on the application of ferric chloride as a pre-treatment step in WWTPs for the control of odours caused by sulphurous gases.

Ferric ions (Fe^{3+}) play a crucial role by chemically oxidizing sulphide to S^0, with a concomitant reduction of Fe^{3+} to Fe^{2+}. This is followed by the precipitation of sulphide as FeS, as shown in Eq. (8) and Eq. (9) (Firer et al., 2008). Due to its high effectiveness, approximately 66% of the total sewage subjected to chemical dosing in Australia is treated with iron salts (Ganigue et al., 2011). Additionally, Fe^{3+} has been reported to inhibit both sulphate-reducing and methanogenic activities in sewer biofilms (Zhang et al., 2009).

$$Fe^{2+} + HS^{-} \rightarrow \downarrow FeS(s) + H^{+} \tag{8}$$

$$2Fe^{3+} + HS^{-} \rightarrow 2Fe^{2+} + \downarrow S^{0}(s) + H^{+} \tag{9}$$

In this context, pH significantly impacts sulphide precipitation, particularly within the sewage pH range of 6–8 (Firer et al., 2008). A low pH significantly increases the Fe demand, which results in high operational costs (Ganigue et al., 2011). More specifically, Firer et al. (2008) demonstrated that when the pH decreased from 8.1 to 6.5, the molar ratio of Fe^{2+} to S^{2-} required to achieve a sulphide concentration of 0.1 mg S/L increased by 3.5-fold. In practical applications, iron salts must be supplied in excess to account for the presence of competing anions in sewage, such as carbonate, which can also react with iron ions. Indeed, a minimal molar ratio Fe:*S* of 1.3:1 for ferrous salts and Fe:*S* of 0.9:1 for ferric salts or a mixture of both salts (at a ratio of Fe^{3+}:Fe^{2+} = 2:1) is required to reduce sulphide below 0.1 mg S/L at pH around 7.0 (Firer et al., 2008). Likewise, the dosage of iron-rich sludge, primarily containing Fe^{3+}, significantly decreased sulphide concentrations from 15.5 to 19.8 mg S/L to 0.7–2.3 mg S/L in a laboratory-scale rising main sewer. This reduction was achieved at a sludge dosing rate that provided a Fe:*S* molar ratio of 1:1, with a sludge to wastewater volume ratio of 0.26% (Sun et al., 2015). This strategy also reduced CH_4 emissions by 20%, which was attributed to the reduction in soluble organics via iron sludge adsorption and precipitation.

Although iron salts have demonstrated a strong capacity to control sulphides and odorous compounds, their use introduces operational challenges such as sludge production and elevated costs under low pH conditions. Research should prioritize developing modified iron salts or dosing protocols that optimize their performance while minimizing environmental and economic impacts.

4.4. Addition of alkali

The addition of alkali to wastewater causes an immediate rise in the pH, thereby reducing the volatilization of H_2S from the liquid phase to the atmosphere. $Mg(OH)_2$ and NaOH are the primary alkalis dosed in sewer systems for this purpose (Gutierrez et al., 2009). $Mg(OH)_2$ is a weak base and a non-hazardous compound extensively used in sewer systems for odour and corrosion control. Due to its low solubility in water (0.009 g/L at 18 °C), $Mg(OH)_2$ can only increase the pH of sewage up to 8.5–9.0. This limited pH increase is typically sufficient to inhibit the release of H_2S and mitigate malodourous emissions without causing significant operational issues in the WWTP. The self-buffering capacity of $Mg(OH)_2$ enables it to maintain residual or unreacted $Mg(OH)_2$ within the sewer system, thereby extending its effectiveness mitigating sulphide emissions over long distances from the initial dosing point. This feature allows for a sustained odour control and corrosion prevention even in areas far downstream of the application site (Gutierrez et al., 2009). The effectiveness of $Mg(OH)_2$ in controlling sulphide emissions is attributed to its dissolution under lower pH conditions, typically below 9.0, due to ongoing re-acidification processes. The increase in the pH up to 8.5–9.0 has been observed to reduce the activity of SRB in sewage by approximately 25–50% (Gutierrez et al., 2009). In laboratory-scale urban wastewater systems, Cen et al. (2023) demonstrated that dosing $Mg(OH)_2$ at a concentration of 56 mg/L conferred multiple advantages on downstream treatment processes: More specifically, $Mg(OH)_2$ dosing increased the sewage pH to 8.8 ± 0.1, decreased sulphide concentrations by 35.1% ± 4.9% (down to levels of 6.7 ± 0.9 mg S/L) and mitigated CH_4 emissions by 58.0% ± 4.9%. In addition, overdosing $Mg(OH)_2$ may result in an excess of unreacted alkalinity in the wastewater, which might be potentially beneficial for downstream nitrification but is not cost-effective (Cen et al., 2023).

While $Mg(OH)_2$ is typically dosed continuously, NaOH is dosed intermittently to cause periodic pH shocks above 10.0 for a short period (hours) (Gutierrez et al., 2009). In controlled laboratory studies, the exposure of sewer biofilms up to a pH of 10.5 for 2–6 h resulted in a 85–90% reduction in SRB activity (Gutierrez et al., 2014). However, SRB re-growth was observed within 1–3 days post-dosing, which reached pre-treatment levels within 3–14 days. Interestingly, field studies conducted in a 1080-m-long rising main sewer revealed that SRB recovery was significantly shorter compared to laboratory conditions (Gutierrez et al., 2014). This discrepancy was attributed to the different pH observed at the end of the pipe (approximately 9.0) due to the dilution effect of the wastewater, which required higher dosages of NaOH upstream to maintain the elevated pH of 10.5 at the end of the sewer pipeline. Therefore, the practical and economic viability of intermittent NaOH dosing needs further optimization. Additionally, the high pH resulting from this shock treatment must be neutralized before discharge into the headworks of the WWTP, which can be achieved through chemical treatment or a storage-release strategy (Gutierrez et al., 2009).

Alkali addition, particularly using $Mg(OH)_2$, offers sustained odour control downstream, but intermittent NaOH dosing faces limitations due to rapid SRB recovery and potential toxicity. Future studies should explore combining alkali treatments with biological methods to extend SRB inhibition and minimize chemical usage.

4.5. Addition of hydrogen peroxide

H_2O_2 has been used to effectively reduce BOD, chemical oxygen demand (COD), offensive odours and foam in domestic or industrial wastewaters (Shepherd and Shreve, 1973). This reagent is a powerful oxidant commonly injected into a wastewater stream (e.g., force main, gravity sewer) to prevent the formation and emission of H_2S and other malodorous compounds associated with wastewater collection and treatment (Ksibi, 2006; Glaze et al., 1990). At pH values higher than 8, the chemical oxidation of sulphur compounds by H_2O_2 results in SO_4^{2-} ions or colloidal sulphur, which does not contribute to the concentration of total BOD or COD. The reaction mechanism following the addition of H_2O_2 into wastewater is the oxidation of sulphides and sulphites into SO_4^{2-} according to the equations below:

$$S^{2-} + 4H_2O_2 \rightarrow SO_4^{2-} + 4H_2O \tag{10}$$

$$SO_3^{2-} + H_2O_2 \rightarrow SO_4^{2-} + H_2O \tag{11}$$

$$HSO_3^{-} + H_2O_2 \rightarrow HSO_4^{-} + H_2O \tag{12}$$

Previous studies have showed that 1–2 mg H_2O_2 are required to remove 1 mg of sulphide in domestic sewers, preventing the formation of odorants and ultimately concrete corrosion (Shepherd and Shreve, 1973). Dębowski et al. (2022) also demonstrated that 2 mL H_2O_2/L supported the oxidation of sulphides into SO_4^{2-} and most organic compounds were destroyed (85% reduction of COD) in municipal wastewater. The odour concentration (OC) of raw wastewater (2980 ± 110 ou_E/m^3) was also reduced by 96.3% ± 1.9% to a level of 100 ± 15 ou_E/m^3. Despite the multiple advantages of H_2O_2 dosing as odour prevention strategy (i.e., low capital costs, fast reaction time, higher dissolved oxygen level of wastewater and no production of harmful by-products), H_2O_2 is considered a hazardous chemical and therefore, requires secondary containment, special operator training, handling and safety precautions.

Although H_2O_2 rapidly oxidizes sulphides and reduces odour concentrations, its hazardous nature poses significant challenges for practical use. Research is needed to explore safer, cost-effective alternatives or synergistic applications with other oxidants for broader implementation.

4.6. Addition of chlorine

Chlorine is the most widely used disinfectant during municipal wastewater treatment because it destroys target pathogenic organisms by oxidizing their cellular material (Otter et al., 2020). Chlorine can be supplied as chlorine gas, hypochlorite solutions and other chlorine compounds in solid or liquid form. The required degree of disinfection can be achieved by varying the dosage and the contact time in the chlorine-based disinfection system. The addition of chlorine during wastewater treatment is an effective strategy for minimizing odours by oxidizing H_2S and other odorous compounds into less odorous substances. It also serves as a disinfectant, reducing microbial populations that can contribute to odour formation. Chlorine dosage during odour mitigation typically varies based on wastewater characteristics and discharge requirements. For instance, Cadena and Peters (1988) assessed sulphide removal by chlorine addition into a wastewater containing 1.4 mg S^{2-}/L and reported total sulphide removal using 4.2 g chlorine per gram of sulphide. A recent lab scale study has reported 20% of sulphide removal in wastewater containing 1.6 mg S^{2-}/L when 15 mg/L of chlorine were added (Freitas et al., 2021). However, no information was provided about the composition of the wastewater used in the study and the occurrence of competing reactions (e.g. organic matter, microorganisms, and ammonia) in order to explain the low sulphide removal recorded.

Chlorine is relatively inexpensive and easy to apply. However, chlorine dosage poses several health challenges, including the formation of harmful disinfection by-products such as trihalomethanes (THMs) and haloacetic acids (HAAs) (Sinha et al., 2021). This reagent is also corrosive, potentially damaging infrastructure, and may not effectively neutralize all odorous compounds, especially complex organics. In addition, safety concerns arise from handling chlorine gas and solutions, and any residual chlorine requires dichlorination before the treated wastewater can be safely discharged into natural water bodies, which ultimately increases treatment complexity and cost.

Chlorine is effective for odour mitigation, but the formation of harmful by-products such as THMs and HAAs limits its environmental viability. Investigating alternative chlorination methods or combining chlorine with advanced oxidation processes could mitigate these drawbacks while maintaining efficacy.

4.7. Addition of potassium permanganate

$KMnO_4$ is a powerful and versatile oxidant that can be used to destroy sulphide, in addition to other organic and inorganic wastewater compounds, over a relatively broad pH range (Wlodarchak et al., 2012). Additional compounds that are effectively oxidized by permanganate include mercaptans and phenols. Under acidic, neutral and alkaline conditions, $KMnO_4$ can react with the H_2S dissolved in the wastewater following equations (13)–(15).

$$3H_2S + 2KMnO_4 \rightarrow 3S^0 + 2KOH + 2MnO_2 + 2H_2O\ (acidic) \tag{13}$$

$$3H_2S + 4KMnO_4 \rightarrow 2K_2SO_4 + S^0 + 3MnO + MnO_2 + 3H_2O\ (neutral) \tag{14}$$

$$3H_2S + 8KMnO_4 \rightarrow 3K_2SO_4 + 2KOH + 8MnO_2 + 3H_2O\ (alkaline) \tag{15}$$

The pH of the $KMnO_4$ solution plays a key role in the mechanisms of these oxidation reactions. From an economic point of view, $KMnO_4$ performs optimally in strongly acidic solutions. However, due to the corrosive nature and chemical instability of such solutions, they are rarely used in practice. On the other hand, highly basic solutions (i.e., pH > 10) cause the reduction of $KMnO_4$ to potassium manganate (K_2MnO_4), a compound with low oxidizing power under the conditions typically used for odour removal in WWTPs. Consequently, the pH for $KMnO_4$ addition is typically maintained in the range of 8.0–9.5 (Medialdea et al., 2005). Overall, several reactions ranging between these extremes may take place yielding not only S^0 and/or SO_4^{2-}, but also thionates, dithionates, and manganese sulphide as potential byproducts, which depends on the reaction conditions (Ficek, 1985). Therefore, the most effective and economical dosage to be applied for odour removal must be empirically determined for each wastewater. A typical dosage of $KMnO_4$ is 6–7 mg per mg of sulphide (Wlodarchak et al., 2012). In addition, a drawback of using this oxidant for H_2S abatement is the production of manganese dioxide (MnO_2), which precipitates as an inert sludge with a brown or black coloration.

In literature, $KMnO_4$ was used during a plant trial to reduce odour emissions in a WWTP in the southeastern of the United States. The authors reported that the level of H_2S recorded within the plant above the belt presses were reduced from 815 to 0 ppm_v after the addition of $KMnO_4$ (Wlodarchak et al., 2012). Similarly, the dosage of $KMnO_4$ at 0.1 g per gram of total suspended solids (TSS) in waste activated sludge induced a reduction in the maximal specific H_2S production from $292.3 \cdot 10^{-4}$ to $96.4 \cdot 10^{-4}$ mg per gram of volatile suspended solids (VSS), while the maximal short-chain fatty acids production increased from 23 to 251 mg COD/g VSS. In fact, experimental results revealed that the addition of 0.1 g $KMnO_4$/g TSS inhibited sulphur-containing organics hydrolysis, amino acids degradation, and sulphate reducing process by 12.5%, 27.2%, and 61.2%, respectively.

The versatility of $KMnO_4$ in odorous compound oxidation is promising, but its operational cost and potential for over-oxidation at higher pH levels restrict its widespread use. Future work should focus on developing controlled dosing systems and understanding its interaction with complex wastewater matrices to improve efficiency.

5. Odour prevention strategies in composting

Composting is a common source of greenhouse gas and odorous emissions (Nordahl et al., 2023). Most published studies in this field focused on the determination of the concentrations of key odorants, particularly H_2S and NH_3, their health impacts, and on the assessment of the efficiency of mitigation strategies (González et al., 2024). In this context, cost-effective odour prevention strategies are critical for ensuring the efficient decomposition of organic matter while minimizing the formation of odorants. These strategies integrate chemical and biological dosing approaches to optimize microbial activity, maintain appropriate moisture levels, and ensure adequate aeration throughout the composting process. Table 4 provides an overview of different odour prevention strategies implemented in composting, highlighting their reaction mechanisms, dosages or application ratios, and effectiveness in reducing odour emissions. In addition, Table 5 summarizes the key features, advantages, and disadvantages of each odour prevention strategy discussed in this section.

5.1. Optimization of the composting conditions (pH and oxygen supply)

Composting conditions govern odorant production and emissions. Basic operational parameters such as O_2, pH and temperature should be in the appropriate range. If composting operation can maintain these conditions under an optimal range, appropriate adjustments to reduce odour emissions are the most convenient and feasible control measures. Temperature is one of the most important factors in the composting process. High temperatures will promote both the diffusion of odours and their formation as a result of the high metabolic activity of the microbial community. However, odour emission prevention by reducing process temperature will entail a pernicious decrease in the compost quality and maturity. Therefore, odour reduction via temperature control is not a feasible strategy.

On the other hand, the O_2 content within the composting pile exerts a significant influence on the generation of odours. Thus, an increase in the oxygen content promotes NH_3 oxidation, thereby reducing the release of NH_3, while the emission of VSCs mainly occurs under insufficient oxygen availability. In this context, oxygen concentration within a compost pile is regulated through aeration, which is a key factor in managing odour emissions. Therefore, an increase in the frequency of aeration or the implementation of continuous aeration to maintain high O_2 levels can effectively reduce the production of VSCs in compost piles. However, this approach is less effective for controlling NH_3 emissions (Chen et al., 2011). In practice, high aeration rates can inadvertently increase the release of gases from the compost pile due to the blow-off effect (Zhao et al., 2019). Indeed, several studies have consistently demonstrated that higher aeration rates are associated with increased NH_3 emissions (de Guardia et al., 2008). In this context, while moderate aeration can decrease VSC emissions, an excessive aeration may actually elevate their release and lead to significant heat loss from the compost, potentially resulting in process failure. In addition, improper aeration can affect not only odour emissions but also the quality of the compostable materials and other gaseous emissions, thereby influencing the overall environmental performance of the process (Blazy et al., 2014). Approaches such as intermittent aeration or advanced aeration control systems, guided by real-time monitoring of oxygen levels, temperature, and emissions, could dynamically adjust aeration rates to maintain optimal composting conditions. Such strategies ensure efficient microbial activity, minimize environmental trade-offs, and enhance the overall sustainability of the composting process. Thus, while aeration is a crucial tool for odour control, balancing its intensity is challenging, and its effectiveness in reducing overall odour generation is limited.

Table 4

Overview of odour prevention strategies in composting: Reaction mechanism, dosage or ratio applied and odour reduction capacity.

Odour prevention strategy	Reaction mechanism	Dosage or ratio applied	Odour reduction capacity	References
O_2 supply	Promotes NH_3 oxidation Inhibits the growth of SRB	Aeration rates from 0.1 to 0.3 L per kg dry matter	Reduction of VSCs and NH_3 by 30.7% and 51.3%, respectively	(Zhang et al., 2016)
Adjustment of pH	Influences the NH_4^+ to NH_3 ratio A high pH promotes NH_3 volatilization A low pH retains H_2S in its ionic forms	Reduction of the pH from 7.5 to 9	Reduction of cumulative NH_3 emissions by 47.8%, but increase the H_2S emissions by 55%	Gu et al. (2018)
Bulking agent	Improves aeration, moisture management, and porosity of compostable materials Essential for efficient decomposition and odour control	Addition of dry cornstalks at a mixing ratio of 4:1 (wet weight) Addition of dry wood chips with eggplant waste at a mixing ratio of 3:1 (wet weight)	Reduction of NH_3 emissions by 30% Reduction of VSCs emissions by 70%	Yang et al. (2013)
Iron salts	$FeCl_3$ promotes coagulation and subsequent NH_3 removal Iron salts can react with dissolved sulfide to form elemental sulphur and sulfates	$FeCl_3$ dosage in the raw materials was 10% of the TN (by molar mass)	Reduction of NH_3 and H_2S emissions by 42% and 76%, respectively	Yuan et al. (2015)
Struvite	Increases the total nitrogen content in the final compost, thus reducing NH_3 emissions Increases the pH of the compost, thus reducing H_2S emissions	$Mg(OH)_2$ and H_3PO_4 dosage were 10% of the TN (by molar mass)	Reduction of NH_3 and H_2S emissions by about 50%	Zhang et al. (2013)
Surfactants	Suppresses odour dissipation to the surrounding environment and distributes the bacterial strain uniformly throughout the compost surface	Spray surfactant on organic waste at a ratio of 3:1)	Reduction in NH_3 emissions, ranging from 85.7% to 100% Reduction in H_2S emissions by 50%	(Gautam et al., 2024; Xi et al., 2005)
Biochar	Provides extensive surface area for the adsorption of gases, thereby reducing the concentration of odours in the composting environment	Addition of biochar at a mixing ratio of 20% w/w	Reduction of NH_3 and VSCs emissions by 64% and 71%, respectively	Steiner et al. (2010)

(continued on next page)

pH is another critical factor influencing odour formation and

Table 4 (*continued*)

Odour prevention strategy	Reaction mechanism	Dosage or ratio applied	Odour reduction capacity	References
Zeolite	Adsorbs excess nitrogen and retains ammonia during the thermophilic stage of composting	Application of 31.5–47.2% zeolite in granular form is the best dose for reducing ammonia emission during composting	Reduction of NH_3 emissions by 20%	Madrini et al. (2016)
Woody peat	Provides high organic content, high cation exchange capacity, and porous nature, thereby reducing the concentration of odours during composting	Addition of woody peat at a mixing ratio of 10–15% w/w	Reduction of NH_3 emissions by 36%	Yuan et al. (2019a)
Medical stone	Provides sponge structure, cation-exchange, special porous properties and large specific surface area	Addition of medical stone at a mixing ratio of 2.5–10% w/w	Reduction of NH_3 emissions by 28–49%	Wang et al. (2016)
Microbial agents	Affect the composting microbial community, or may inhibit odour-causing microorganisms	Inoculate 5% of laboratory-preserved strain Thiobacillus thioparus 1904	Reduction of NH_3 emissions by 21.8%, Reduction of cumulative H_2S, DMS, MM and DMDS emissions of by 33.2%, 81.2%, 32.7% and 54.2%, respectively	Gu et al. (2018)
Mature compost	Shares several advantages with other odour mitigation strategies, such as bulking agents, adsorbents, and microbial inoculants	Addition of mature compost at a mixing ratio of 10% w/w	Reduction of NH_3 and H_2S emissions by 58.0% and 65.1%, respectively	Yang et al. (2019)

emission during composting, which fluctuates as the composting process progresses. In the initial stages of composting, the decomposition of organic matter entails the release of organic acids, causing a drop in pH. As composting continues, these acids are further broken down, and sulphur-containing compounds decompose, generating sulphide ions. These ions combine with hydrogen ions (H^+), thus reducing the overall H^+ concentration and resulting in an increase in pH. Additionally, the release of NH_3 generated from the mineralization of organic nitrogen also contributes to a rise in pH. However, nitrification reactions also occur at the end of the composting process, thus releasing H^+ ions, which can induce a further decrease in pH (Lin, 2008). In this context, pH plays a key role in the ratio of NH_4^+ to NH_3 during composting. Liang et al. (2004) simulated the volatilization of NH_3 during composting and confirmed that significant NH_3 volatilization occurs at high pH levels. In contrast, H_2S is less volatile under alkaline conditions. Thus, an alkaline pH helps retaining H_2S in its ionic forms, reducing its emissions and even facilitating its absorption. Finally, while lowering the pH of compost can reduce NH_3 production, it often leads to increased emissions of VSCs. In fact, Gu et al. (2018) found that reducing the pH in composting decreased cumulative NH_3 emissions and total nitrogen losses by 47.8% and 44.2%, respectively, but also increased VSC emissions and total sulphur losses. Therefore, pH control as a strategy to mitigate odour emissions is complex and typically not effective.

Current research demonstrates that controlling composting conditions like oxygen content and pH is critical for odour management. However, these strategies often involve trade-offs. For example, while increasing oxygen supply reduces VSC emissions, it can inadvertently exacerbate NH_3 emissions due to the blow-off effect. Similarly, adjusting pH to mitigate NH_3 emissions may enhance VSC release. Studies largely neglect the synergistic effects of optimizing multiple parameters dynamically, particularly through real-time monitoring systems. Advanced aeration technologies, coupled with AI-driven monitoring tools, could provide tailored solutions to balance these competing factors, enhancing overall odour control without compromising compost quality.

5.2. *Use of additives*

Composting additives represent a promising alternative for odour control. These additives typically have a minimal impact on the overall environmental conditions of the composting process but can effectively reduce odours through various mechanisms such as porosity enhancement and odorant adsorption. Odour-preventing additives in composting include bulking agents, chemical agents, adsorbents, microbial inoculants, and mature compost, among others. These additives help creating a more stable composting environment and reducing the risk of odour emissions, while supporting an effective composting process.

5.2.1. *Addition of bulking agents*

Bulking agents are materials added to composting piles to improve critical parameter for minimizing odour production such as aeration, moisture management, and porosity. These agents, typically composed of carbon-rich dry materials such as straw, wood chips, cornstalks, or shredded branches, help creating a more open structure within the compost pile. This porous structure enhances airflow, thereby maintaining the aerobic conditions that are essential for efficient decomposition and odour control (Guidoni et al., 2018). Odours are often generated when anaerobic conditions prevail, leading to the production of compounds such as H_2S and VOCs. By increasing the porosity of the compost, bulking agents facilitate oxygen penetration throughout the pile, which supports aerobic microbial activity and prevents the formation of malodorous compounds.

For instance, cornstalks have been successfully tested as a composting bulking agent in literature (Guo et al., 2012). Yuan et al. (2015) reported that the addition of cornstalks can reduce NH_3 and VSC emissions by 30.5% and 70%, respectively. In fact, cornstalks can absorb a significant amount of NH_4^+/NH_3, avoid the formation of leachate and reduce nitrogen losses from leachate. However, other studies have shown that the NH_3 emission mitigation capacity is limited to ~6% since the addition of cornstalks increases the pH and the aeration of the substrate. This ultimately accelerates the decomposition and conversion of organic matter and its effect on reducing NH_3 emissions is not significant (Yang et al., 2013).

The addition of wood chips with a small fraction of aerial eggplant waste into a full-scale plant composting sewage sludge significantly reduced odour concentrations compared to a reference compost pile. The composting pile with wood chips and eggplant waste emitted an odour concentration of 6130 ou_E/m^3, which was significantly lower than the 22500 ou_E/m^3 observed in the reference pile (Toledo et al., 2019b). On the other hand, Li et al. (2017) reported that mushroom substrate, due to its lower pH, exerted a higher mitigation of NH_3

Table 5
Key features, advantages, and disadvantages of each odour prevention strategy in composting plants.

Treatment	Key Features	Advantages	Disadvantages	Reference
Optimization of pH	pH influences NH_3 volatilization and the retention of sulphur compounds	Alkaline pH retains H_2S as ionic forms Mitigates NH_3 emissions by lowering pH	Alkaline pH increases NH_3 volatilization Acidic pH increases VSC emissions Complex to balance effectively	(Gu et al., 2018; Liang et al., 2004)
Oxygen supply	Aeration regulates O_2 levels and maintains aerobic conditions	Promotes NH_3 oxidation Reduces anaerobic VSC emissions Supports aerobic microbial activity	Excessive aeration increases NH_3 blow-off and heat loss Challenging to balance for optimal results	(de Guardia et al., 2008; Zhao et al., 2019)
Addition of bulking agents	Use of carbon-rich materials (e.g., straw, wood chips) to enhance porosity, moisture, and aeration	Reduces NH_3 and VSC emissions Enhances aerobic conditions Cost-effective	Limited mitigation of NH_3 in high pH conditions. Bulking agent type affects efficiency	(Guidoni et al., 2018; Yuan et al., 2015)
Addition of chemical agents	Application of $FeCl_3$, $Mg(OH)_2$, phosphoric acid, and surfactants to reduce emissions through chemical reactions	Effective for NH_3 and H_2S reduction Can preserve nitrogen (e.g., struvite formation) Quick results	High cost Repeated applications needed Requires optimization for cost-effectiveness	(Ren et al., 2010; Yuan et al., 2015)
Use of adsorbents	Porous materials (e.g., biochar, zeolite, peat) adsorb odorous gases	Reduces NH_3 and VSC emissions Supports microbial activity Reusable and sustainable (e.g., biochar)	Initial cost for high-quality adsorbents Efficiency varies with material type and composting conditions	(Sanchez-Monedero et al., 2018; Yuan et al., 2019a)
Inoculation with microbial agents	Introduction of microbes (e.g., Bacillus, ammonia-oxidizing bacteria) to degrade odorants or inhibit odour-producing pathways	Synergistic effects of mixed microbes Tailored microbial consortia improve effectiveness Promotes odour reduction biologically	Complex optimization required for microbial selection Limited performance in highly variable composting conditions	(Duan et al., 2020; Sánchez et al., 2017; Xu et al., 2022)
Use of mature compost	Application of mature compost as a cover or additive	Reduces NH_3 and VSC emissions Acts as a bulking agent and microbial inoculant Cost-effective	Effectiveness varies with moisture and organic content Requires further optimization for specific scenarios	(Hort et al., 2013; Kato and Miura, 2008)

emissions (50% reductions) compared to cornstalks (30% reductions). However, the mushroom substrate was less effective than cornstalks adsorbing VSCs, with a reduction of 72% compared to 80%. Finally, sawdust, which is characterized by its lightweight and fine particle size, can create a powder-like consistency that coats the surface of composting materials effectively and enhance its ability to adsorb malodours efficiently.

The use of bulking agents such as wood chips, cornstalks, and mushroom substrates effectively reduces odour emissions, but their performance varies significantly depending on the compost matrix and conditions. Research has focused on their immediate impact, often ignoring the potential environmental costs of their production and application. Future investigations should prioritize lifecycle analyses to ensure these materials provide a net environmental benefit. Moreover, combining different bulking agents to leverage their complementary properties could optimize their effectiveness.

5.2.2. *Addition of chemical agents*

The addition of chemical agents to the compost matrix can mitigate odours via specific chemical reactions. More specifically, $FeCl_3$ is commonly used to remove NH_3 from wastewater, as discussed in section 4.3, and has been also applied to *in-situ* control VSCs and H_2S in anaerobic digestion. Although the use of iron salts in composting is less common, studies have demonstrated the feasibility of $FeCl_3$ in odour reduction during composting. For instance, Yuan et al. (2015) observed that the addition of $FeCl_3$ to compost reduced NH_3 and H_2S emissions by 42% and 76% compared to control tests, respectively. This NH_3 reduction was attributed to the potential of $FeCl_3$ as a flocculant, which promotes coagulation and subsequent NH_3 removal. In the case of H_2S, iron salts react with dissolved sulfides through various pathways, forming elemental sulphur and SO_4^{2-}.

On the other hand, struvite ($NH_4MgPO_4{\cdot}6H_2O$) crystallization has emerged as an effective strategy for nitrogen conservation during composting. By adding magnesium and phosphorus salts to composting materials, NH_4^+ can be preserved in the form of struvite (Li et al., 2018). Indeed, the incorporation of $Mg(OH)_2$ and phosphoric acid (H_3PO_4) into pig manure composting increased the total nitrogen content in the final compost and reduced NH_3 emissions by approximately 50% (Ren et al., 2010). This reduction occurred as a result of the combination of phosphate ions with NH_4^+ and Mg^{2+} to form stable complexes of $NH_4MgPO_4{\cdot}6H_2O$ and NH_4CaPO_4, thus inhibiting the conversion of NH_4^+-N into gaseous NH_3 and preventing odour emissions. In addition, compost supplementation with $Mg(OH)_2$ and H_3PO_4 can also mitigate VSC emissions. Zhang et al. (2013) reported that the addition of these salts reduced H_2S emissions by nearly 50% likely due to the increase in the pH of the compost, which plays a key role in VSCs mitigation.

Finally, some surfactants have been explored for their potential to reduce odours during composting, although their development is still in an embryonic stage. A recent field-scale study by Gautam et al. (2024) tested a combination of stable surfactant foam and odour-reducing bacterial strains, which were applied directly to compostable materials to enhance odour degradation and mitigation. This approach resulted in a significant reduction in NH_3 emissions (86–100%) within 24–48 h following sequential surfactant applications on swine manure piles. Likewise, Xi et al. (2005) reported that bio-surfactant addition in the composting of municipal solid waste increased the content of humic substances from 10.6% to 18.2%, and reduced by 50% H_2S emissions.

Overall, chemical dosage for the removal of malodours during composting is often associated with the high costs of chemical purchase and the need for repeated application to maintain effectiveness. In this context, a combined approach that integrates chemical addition with alternative treatment strategies can enhance the cost-effectiveness of odour prevention during composting. For instance, the addition of chemical reagents as a pretreatment to enhance the efficiency of microbial inoculants can significantly improve odour control. This synergistic approach supports a reduction in the dosage of chemical reagents while leveraging the biological activity of microorganisms to further degrade odorous compounds. Thus, the optimization of the dosage and timing of application of both chemical and biological additives will boost the effectiveness and economic feasibility of the active odour reduction during composting.

Chemical agents, such as $FeCl_3$ and struvite-forming salts, show promise in mitigating NH_3 and VSC emissions. However, their high cost and the necessity for repeated applications limit their practical applicability. Additionally, many studies do not assess their long-term impact on compost quality and soil health when applied in agricultural contexts. Future research should focus on integrating chemical agents with biological strategies, such as microbial inoculants, to reduce costs and improve efficacy sustainably.

5.2.3. *Use of adsorbents*

Natural and synthetic adsorbents with a porous structure and high surface area are highly effective capturing and retaining significant amounts of odorants generated during composting. These adsorbents provide a large surface area for the adsorption of gases, thereby reducing the concentration of odours in the composting environment. Furthermore, many adsorbents can be recycled and reused at the end of the composting process, thus contributing to both the economic and environmental sustainability of odour abatement. Zeolite, which is known for its excellent cation-exchange capacity and affinity for ammonia, is a very common adsorbent tested in composting piles (Madrini et al., 2016). Similarly, biochar provides a large surface area and functional groups for the adsorption of a variety of odorous compounds (Sanchez-Monedero et al., 2018). Woody peat is also effective in odour mitigation due to its high organic content and porous nature (Yuan et al., 2019a). Finally, medical stone can effectively adsorb sulphur-containing compounds and other malodorous gases during composting (Wang et al., 2016). Each of these adsorbents offers distinct advantages depending on the specific composting conditions and target odorous compounds.

Among multiple sorbents used in composting for odorant capture, biochar has received most attention due to its effectiveness in reducing odour emissions and retaining nutrients. Studies by Steiner et al. (2010) demonstrated that the incorporation of biochar into compost significantly decreased the concentration of NH_4^+ in the compost, leading to a reduction in NH_3 volatilization by approximately 64% and a decrease in nitrogen loss by up to 52%. This was attributed to the high ion exchange capacity of biochar, which allow it to capture large quantities of NH_4^+. Moreover, biochar supports the growth of nitrifying microorganisms, which can reduce N_2O emissions and promote the conversion of ammonia to nitrate, thereby retaining nitrogen in the composting process (Chen et al., 2017). In addition to it potential to reduce nitrogen-related emissions, biochar has also been shown to decrease VSC emissions by up to 71% (Steiner et al., 2010). The primary mechanism for this reduction is likely the improvement in substrate aeration facilitated by the porous structure of biochar, which enhances microbial activity and prevents the occurrence of the anaerobic conditions that promote VSC formation. Biochar also exhibits strong adsorption affinity for SO_4^{2-}, which can influence the microbial community dynamics within the compost. However, further research is needed to fully understand the interactions between the physical and chemical properties of biochar, and the specific mechanisms underlying the mitigation of VSC emissions in composting.

Natural adsorbents like biochar, zeolite, and woody peat have demonstrated high effectiveness in odour control. Despite promising results, the variability in adsorbent quality and the lack of standardization in application methods pose challenges for scaling up their use. Future research should explore tailoring adsorbent properties to specific composting conditions and odorants. Moreover, the potential for recycling and reusing adsorbents at the end of the composting process requires further investigation to maximize their environmental and economic benefits.

5.2.4. *Inoculation with microbial agents*

Beyond their primary adsorption capabilities, certain adsorbents can also influence the microbial communities within the composting matrix, thereby contributing an indirect microbial-based reduction of malodorous emissions. Modifying the microbial composition of composting piles can effectively decrease the formation of odorants by regulating microbial metabolic pathways. To directly regulate the microbial communities responsible for odour generation, the use of microbial inoculants or metabolic inhibitors targeting specific metabolic pathways has proven to be more effective. Numerous studies have explored the application of microbial agents such as lactic acid bacteria, *Bacillus* species, and *Saccharomyces*, among others, to control odours during composting processes (Sánchez et al., 2017). These microbial agents either outcompete odour-producing microbes or metabolize their odorous metabolites, thereby minimizing their emissions.

Given the broad diversity of microorganisms, a wide portfolio of microbial inoculants can be introduced into composting systems. Some microbes directly interact with nitrogen and sulphur compounds, altering their transformations, while others inhibit key microbial species responsible for odour production. For example, *Thiobacillus thioparus*, a type of sulphur-oxidizing bacteria, has been widely used for odour control in biofiltration systems, though it is less common in aerobic composting processes. When introduced into composting piles, *T. thioparus* significantly decreased the cumulative emissions of H_2S, DMS, MM, and DMDS by 33.2%, 81.2%, 32.7%, and 54.2%, respectively (Gu et al., 2018). This bacterium facilitates the conversion of organic sulphur and elemental sulphur into SO_4^{2-}, thereby enhancing the availability of sulphur in the compost.

Compared to single microbial strains, bioaugmentation with mixed microbial consortia or commercially formulated compost-specific bacteria typically yields more effective results in terms of odour mitigation during composting. Mixed strains, such as a combination of *Bacillus* species and ammonia-oxidizing bacteria, offer synergistic interactions that enhance their ability to degrade or transform odorous compounds. For instance, *Bacillus* species can rapidly decompose organic matter, reducing the availability of substrates available for the production of odorous VOCs (Duan et al., 2020), while ammonia-oxidizing bacteria convert NH_3 into less volatile forms such as nitrites NO_2^- and NO_3^-, thereby minimizing NH_3 emissions (Xu et al., 2022). The mechanisms underlying odour suppression by these mixed microbial communities are more complex and multifaceted compared to single-strain inoculants. Thus, mixed microbial communities typically involve a combination of biochemical transformations, competitive inhibition, and environmental modifications. For example, certain bacterial strains may lower the pH of the compost matrix, thus reducing the volatility of NH_3, while others produce enzymes that specifically break down VSCs such as H_2S and DMS (Lin et al., 2022). The effectiveness of mixed microbial communities in odour control is highly dependent on the careful selection of microbial species and their ratios, which must be tailored to the specific composting conditions and targeted odorous compounds. This process involves optimizing parameters such as the metabolic capabilities of the microbes, their growth rates, the nature of the organic waste, moisture levels, and temperature. Additionally, the compatibility of different microbial strains must be considered to avoid antagonistic interactions that could reduce their collective effectiveness. Recent advances in microbial ecology and biotechnology, such as metagenomic analysis and metabolic profiling, have greatly improved our ability to design and optimize these microbial consortia for specific odour control applications in composting, leading to more predictable and reliable outcomes (Zhou et al., 2024).

Microbial inoculants, including *Bacillus* species and ammonia-oxidizing bacteria, have shown significant potential in odour mitigation through competitive inhibition and biochemical transformations. However, the effectiveness of microbial consortia is highly dependent on specific composting conditions, and there is limited understanding of the interactions between microbial communities and compost substrates. Advances in microbial ecology, such as metagenomics and metabolic profiling, offer opportunities to design customized microbial solutions tailored to specific odour control challenges.

5.2.5. Use of mature compost

Mature compost, the end product of the composting process, shares several advantages with other odour mitigation additives such as bulking agents, adsorbents, and microbial inoculants. The supplementation of mature compost has been suggested as a viable method for controlling odour emissions during composting due to its readily available nature, porous structure, rich microbial content, and cost-effectiveness. Currently, mature compost is widely used for odour control in biological filters, where its high porosity and microbial diversity help in the adsorption and biodegradation of odorous compounds such as NH_3, VSCs and VOCs (Hort et al., 2013). However, its effectiveness in the mitigation of odour emissions during the composting process itself remains debated, particularly when odour control is more challenging due to higher moisture content and organic load. Some studies have shown that applying a layer of mature compost directly on top of composting piles can significantly reduce NH_3 emissions, primarily through physical adsorption and microbial assimilation, with a concomitant increase in NH_4^+ accumulation within the covered material (Luo et al., 2014). Yang et al. (2019) reported that mixing mature compost into composting materials reduced NH_3 emissions by 58.0%. Thus, mixing mature compost may be a potential effective ammonia control tool. The effectiveness of mature compost can be attributed to its function as a bulking agent, which improves inter-particle voids in the compost pile. This enhancement of air permeability promotes aerobic conditions, thereby optimizing microbial activity and reducing the generation of anaerobic odours. Additionally, mature compost has a high surface area due to its porous structure, which enhances its adsorption capacity for various odorous compounds.

Furthermore, the rich microbial community of mature compost plays a key role in odour mitigation by facilitating the rapid succession and stabilization of beneficial microbial populations, which can accelerate the composting process and reduce the duration of odour emissions. The microbial diversity in mature compost can also create a more conducive environment for microbial growth within the composting piles, promoting the breakdown of odorous compounds more effectively than in immature compost environments. Therefore, the use of mature compost as both a bulking agent and a microbial inoculant entails a multiple approach for enhancing organic matter stabilization and reducing odour emissions (Kato and Miura, 2008). However, further research is needed to fully understand its mechanisms of action and optimize its dosage and application frequency, particularly in challenging composting scenarios such as those involving different compostable materials. On the other hand, the mechanism by which mature compost inhibits VSCs is similar to its effect on NH_3 emissions. Several factors, including adsorption, the promotion of beneficial microbial activity, and the adjustment of matrix porosity and moisture content, contribute to the reduction of VSC emissions. For instance, Yuan et al. (2019b) demonstrated that covering a compost pile with mature compost resulted in a 65.1% reduction in H_2S emissions, which was partially due to the inhibition of SRB. Additionally, recent works suggest that mature compost promotes the growth of denitrifying bacteria, which in turn inhibits the proliferation of SRBs, further contributing to the reduction of VSC emissions (Wang et al., 2018). This multifaceted approach highlights the complex interactions between physical, chemical, and biological factors during the management of malodours in composting.

Mature compost offers a dual function as a physical and biological odour mitigator, yet its mechanisms of action remain underexplored. Current studies focus primarily on its immediate adsorption and microbial assimilation capabilities, with limited insights into its long-term impact on compost quality. Further research should address the optimization of mature compost application rates and methods, particularly in high moisture, high organic load environments where odour control is most challenging.

6. Conclusions and future prospects

Unpleasant and malodorous emissions have become a pressing concern in wastewater treatment and composting facilities, significantly affecting both human health and the environment. Among the wide array of odorants, NH_3 and H_2S are the most prevalent, substantially influencing the odour footprint of WWTPs and CPs. A broad portfolio of strategies has been investigated to prevent odour generation and release in WWTPs and CPs, including the optimization of operational parameters, the application of chemical additives, microbial inoculation, and the use of adsorbents and bulking agents. In this context, chemical and biological dosing have proven to be effective odour prevention strategies. However, the mechanisms underlying odour reduction by chemical and biological dosing are complex and not fully understood, which requires further research to elucidate the biochemical and microbial pathways involved in the degradation of key odorants, such as NH_3 and H_2S. Understanding the dynamics of microbial communities and their interactions with odorants is critical to optimizing biological and chemical dosing. Furthermore, an integrated approach combining multiple odour prevention techniques, tailored to the specific biochemical pathways and operational conditions of WWTPs and CPs, will help developing more effective and cost-efficient odour management solutions. This holistic approach will help minimizing the environmental and health impacts of odour emissions, while improving community relations and reducing operational costs. Ultimately, the insights and strategies discussed in this review provide a solid foundation for the development of sustainable, efficient, and scalable odour management solutions.

CRediT authorship contribution statement

M. Toledo: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Investigation, Conceptualization. **R. Muñoz:** Writing – review & editing, Visualization, Validation, Supervision, Funding acquisition, Conceptualization.

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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Data availability

No data was used for the research described in the article.

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