ANSWERS TO QUESTIONS FROM PANEL
TO DR RISSMAN

1. The Hearing Panel understands that the Ministry for the Environment guideline level for H₂S has for many years been set at 7 µg/m³ in non-geothermally influenced areas, that is to say ten times the level of odour detection (as per Dr Rissmann's evidence). Is there any data as to actual measured levels of H₂S received beyond the boundary of the landfill site relative to guideline levels?

To my knowledge, no measures of hydrogen sulfide or other odour causing gases have been made within the buffer zone of the landfill. Such sampling is possible, but given climatic controls over odour strength, a simple one-off sampling programme is unlikely to provide any rigour as to the likely concentrations[1,2]. A scientifically robust sampling programme would require a long period of relatively high frequency (daily) monitoring over three or more winters if it were to provide any meaningful insight over the concentration of odour, causing gases. The need for high-frequency monitoring is due to climatic variation, which, as noted in earlier evidence, is the primary driver over odour accumulation, transport and sensitivity at Victoria Flats Landfill[1,2]. Further, any consideration of odour sensitivity needs to be cognisant that the active face of the landfill will get closer and closer to the buffer zone over the consented lifetime of the landfill.

As such, any monitoring programme would be complicated, expensive and potentially crude relative to the recognised sensitivity of the human nose and best practice recommendations for the use of buffer zone areas around landfill sites to mitigate odour complaints[1,2]. Please note as part of this evidence the recent work of:

- Ko et al. (2015): Emissions and Control of Hydrogen Sulfide at Landfills: A Review. Critical Reviews in Environmental Science and Technology, and;

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[1] Please note that recent odour impact assessment research showed that atmospheric stability could increase the odour impact radius around landfills between 340 and 1100 percent depending on the land use characteristics of the surrounding areas (Tansel, B., & Inanloo, B., 2019). Odor impact zones around landfills: Delineation based on atmospheric conditions and land use characteristics. Waste Management, 88, 39–47. https://doi.org/10.1016/j.wasman.2019.03.028.

[2] Ko, J. H., Xu, Q., & Jang, Y. C. (2015). Emissions and Control of Hydrogen Sulfide at Landfills: A Review. Critical Reviews in Environmental Science and Technology, 45(19), 2043–2083. https://doi.org/10.1080/10643389.2015.1010427

- Tansel and Inanloo (2019): Odor impact zones around landfills: Delineation based on atmospheric conditions and land use characteristics. Journal of Waste Management.

Both of these journal articles discuss the complexity of odour mitigation and note the importance of maintaining buffer zones to mitigate odour complaints – especially in sensitive climatic settings. The findings of these studies are also consistent with the International Association of Solid Waste and the MfE's best practice guidelines for landfill[3]. However, I note that both scientific papers give greater regard to climatic and other site-specific factors than current regulations, which are generic by necessity. Copies of these manuscripts are provided along with this response.

2. What numerical level of received hydrogen sulphide would Dr Rissmann consider would breach the landfill resource consent requirement to avoid offensive or objectionable odours beyond the boundary of this particular site?

Beyond the boundary of this particular site, a concentration ≥ 7 µg/m^3 would be non-compliant. However, I note that odour complaints may be insensitive to the MfE guideline level for H_2S. For example, a landfill may be compliant with the regulatory threshold but still receive complaints given the high sensitivity of the human nose to odour, causing gases[1,2]. [Members of the public do not know that the concentration of H_2S is compliant]. A high level of odour complaint is especially true of landfill sites characterised by still air conditions, as exemplified by Central Otago winter and the topographic setting of the Victoria Flats Landfill[1].

Recent odour impact assessment research suggests that atmospheric stability (e.g. temperature inversion) can increase the odour impact radius around landfills dramatically (330 – 1,100%; in examples from Tansel, and Inanloo, 2019), depending on the land use characteristics of the surrounding areas[1]. [If the Victoria Flats Landfill were located at the eastern coast, I would be far less concerned, about the potential risk]. As such, it is not merely a matter of odour concentration but also climatic conditions, especially those known to generate temperature inversion that leads to an increased sensitivity and associated frequency of odour complaints.

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[3] International Solid Waste Association - Landfill Operational Guidelines, (2010).

3. We have heard evidence regarding the proposal to capture methane from the landfill. Ignoring any commissioning-related spikes, what difference will that make to odour received beyond the site boundaries, and therefore to the reverse sensitivity risks identified in Dr Rissmann's evidence?

Gas reticulation and flaring is known to reduce overall odour from a site. However, it is often the site of active landfilling and relatively young (0 – 3 year), sulphur-rich waste that is the primary source of odour causing gases. As waste ages, its odour generating potential decreases, in response to the conversion of inorganic and organic sulphur to hydrogen sulfide and other sulphurous gases. Odour generation potential is, therefore exhausted over time. However, 'fresh' waste is brought in each day and disposed of across the area of active landfilling, resupplying the landfill with young sulfur-rich waste. I note two examples of regional landfills with active landfill gas extraction and flaring, Green Island (Dunedin City) and ABLime (Southland) both of which continue to generate odour-causing gases and receive odour complaints.

To my knowledge, having consulted to both landfills, the primary odour causing areas were/are associated with young waste across areas of active/recent landfilling. The area of an active landfilling is not capped and as such less treatable by gas extraction technologies. Further, it is critical to note that neither of these landfills is associated with the same climatic sensitivity as Victoria Flats.

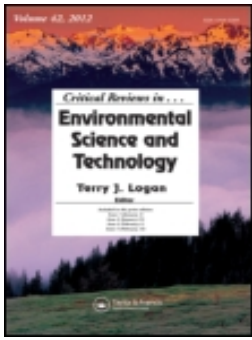*4. Dr Rissmann says at 5.3 of his EIC*

"Due to the potential for detectable odour being exacerbated during wintertime temperature inversions, I cannot recommend any intensive people-related activities (industrial) go into this locality [the buffer area] while the landfill is still in operation".

Mr Giddens for CCCL has offered a new policy and rule prohibiting residential, visitor accommodation, commercial recreation, recreation, and community activities in the buffer area. Does this address Dr Rissmann's concerns?

- Mr Gidden's suggestion does not address my concerns. While the removal of these more sensitive activities would reduce the potential for reverse sensitivity complaints, the level of development still enabled by the GIZ would result in pretty intensive use of the zone, by workers and visitors (2,784[4] daily). The buffer is an important mitigating factor in terms of odour where my advice is that is necessary to retain its current rural / agricultural use. .

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[4] The Buffer Area is 23.2% by land area of the re-zoning's total. The total vehicles per day is agreed by traffic experts to be 24,000 which equates to 5,568 movements or at least 2,784 drivers to the Buffer Area each day.

Emissions and Control of Hydrogen Sulfide at Landfills: A Review

Jae Hac Ko, Qiyong Xu & Yong-Chul Jang

Emissions and Control of Hydrogen Sulfide at Landfills: A Review

JAE HAC KO,[1] QIYONG XU,[1] and YONG-CHUL JANG[2]

[1]Key Laboratory for Eco-Efficient Polysilicate Materials, School of Environment and Energy, Peking University Shenzhen Graduate School, Guangdong, China

[2]Department of Environmental Engineering, Chungnam National University, Daejeon, South Korea

Hydrogen sulfide (H_2S) in landfill gas, formed with the biodegradation of municipal solid waste, is a major odorous component in a landfill. It poses a potential risk to humans and causes odor problems and complaints by the residents near landfills. Many studies have been conducted on landfill gas qualities. Also, various H_2S control technologies have been used in different industrial sectors. However, comprehensive reviews on H_2S issues in landfill gas are rare. An understanding of the formation of H_2S encountered in landfill gas and the emission at which it occurs helps assess risk and problems. The authors review recent publications from different perspectives on H_2S odor in landfills, including H_2S generation the impact on the environment and human health, H_2S levels in the landfill environment, and H_2S control technologies at landfills.

KEY WORDS: hydrogen sulfide, landfills, odor emission, control technologies, environmental and health impacts

1. INTRODUCTION

Landfills are one of the major sources of odorous nuisance in the environment and can cause many complaints from residents living near the facilities (Sadowska-Rociek et al., 2009; Dincer and Muezzinoglu, 2006). To a large extent, the odor problem became one of the main reasons for public opposition

Address correspondence to Qiyong Xu, Key Laboratory for Eco-Efficient Polysilicate Materials, School of Environment and Energy, Peking University Shenzhen Graduate School, University Town, E 118 Building, Nanshan District, Shenzhen, 518055, P. R. China. E-mail: qiyongxu@pkusz.edu.cn

to locating landfill sites (Asakura et al., 2010). The emission of odorous gases from landfills mainly results from the generation of gaseous compounds during the biodegradation of landfilled waste (Dincer and Muezzinoglu, 2006; ElFadel et al., 1997). More than 100 odorous compounds have been identified as contributors to landfill odors, including hydrogen sulfide (H_2S), methyl mercaptan (CH_3SH), dimethyl sulfide ((CH_3)$_2$S), and ammonia (NH_3; He et al., 2012; Kim et al., 2006b; Ying et al., 2012; Kim, 2006). Among the sulfur compounds, H_2S has been identified as a major contributor to odors at landfills (Kim et al., 2006a; Lee et al., 2006; Kim et al., 2005).

The emission of H_2S from landfills has become a growing environmental concern (He et al.et al., 2012; Plaza et al.et al., 2007; Xu et al., 2010b; Sironi et al., 2005). H_2S is a poisonous, flammable, colorless gas with a characteristic odor of rotten eggs. It is known that H_2S poses flammability in air at concentrations between 4.3–45.5% by volume. The odor threshold of H_2S ranges from 0.0005 to 0.3 ppm (Agency for Toxic Substances and Disease Registry, 2008). The average odor threshold, which is defined as detection by 50% of the population, is reported to be 7–9 parts per billion (ppb; Ruth, 1986). Because of the low threshold, landfill gas emitted with a low concentration of H_2S can cause odor complaints by neighbored residents. Not only for odor control but also for landfill gas-to-energy projects, H_2S must be removed from the gas because of its corrosiveness to equipment (Fairweather and Barlaz, 1998). Also, H_2S poses a potential health and safety threat to people living or working near facilities releasing the compound (Ying et al., 2012; Li et al., 2008).

Although many studies have been conducted to control H_2S in off-gases from various sources, few comprehensive reviews have been done on landfill H_2S emission and control. To address H_2S odor problems in a landfill, it is necessary to understand H_2S generation and to manage the landfill properly. This paper reviews literature on H_2S in landfills, including H_2S generation, the risk of H_2S exposure, H_2S emission from landfills, and H_2S control technologies for landfills.

2. H_2S GENERATION IN LANDFILLS

2.1   Generation of H_2S by Sulfate-Reducing Bacteria

Microorganisms are able to produce H_2S from inorganic sulfur compounds, sulfate and sulfite, through the sulfate assimilation pathway (Thomas and Surdin-Kerjan, 1997) or organic sulfur compounds, and through the catabolism of cysteine and glutathione (Landaud et al., 2008; Winter and Curtin, 2012). Hydrogen sulfide can be produced from both bacterial reduction of sulfate and decomposition of sulfur-containing organic constituents under anaerobic conditions in landfills.

The generation of H_2S from dissimilatory sulfate reduction is carried out by a specialized group of anaerobes, called sulfate-reducing bacteria

(SRB). SRB are widely spread in sewer systems, wastewater sludge, and landfills (Hao, 2003). The population of SRB in a lab-scale simulated landfill reached 8.1×10^6 cells per day (Fairweather and Barlaz, 1998). During the dissimilatory reduction process, SRB obtain energy for growth by oxidation of organic substrates and use sulfate as the terminal electron acceptor as Equation 1 (Liamleam and Annachhatre, 2007):

$$SO_4^{2-} + 2C + 2H_2O \xrightarrow{SRB} H_2S + 2HCO_3^- \qquad (1)$$
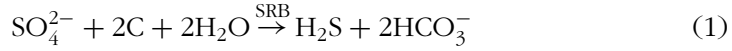
The generation of H_2S through biological sulfate reduction by SRB mainly include three steps: sulfate activation (Equation 2), sulfate reduction to sulfite (Equation 3) and sulfide formation (Equation 4; Barton and Plunkett, 2002):

$$SO_4^{2-} + ATP \xrightarrow{APS\,sulfurylase} APS + PP \qquad (2)$$

$$SO_4^{2-} + ATP + H_2 \rightarrow HSO_3^- + AMP + H^+ \qquad (3)$$

$$HSO_3^- + 3H_2 \rightarrow HS^- + 3H_2O \qquad (4)$$
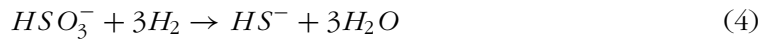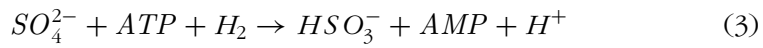
SRB are obligate anaerobic bacteria and can only use sulfate in the absence of electron acceptors such as O_2, H_2O_2, Cl_2, NO_3^-, and Fe^{3+}. Postgate (1984) reported that in order to cultivate SRB, the redox potential of the environment must start around −100 mV (Postgate, 1984). The optimal ORP for H_2S generation was reported in the range of −100 mV to −50 mV (Boon, 1995). SRB prefer an environment around a pH of 7 and H_2S generation are usually inhibited at pH values lower than 5.5 or higher than 9 (Hao, 2003).

In addition to sulfate, most SRB can use sulfite or thiosulfate as electron acceptors. In the presence of thiosulfate or sulfite, SRB first disproportionates the thiosulfate or sulfite compounds to sulfate and then oxidizes the organic substrates with the newly formed sulfate as shown in Equations 5 and 6 (Li et al., 2008).

$$S_2O_3^{2-} + H_2O \xrightarrow{SRB} SO_4^{2-} + HS^- + H^+ \qquad (5)$$

$$4SO_3^{2-} + H^+ \xrightarrow{SRB} 3SO_4^{2-} + HS^- \qquad (6)$$

H_2S is also one of the end products resulting from the action of many varieties of bacteria on organic material containing protein, such as food waste. Amino acids containing sulfur, such as, cysteine (HS-CH_2-CH(NH_2)-COOH) and methionine (CH_3-S-CH_2-CH-CH(NH_2)-COOH) are recognized as the chief sources of H_2S formed during the putrefaction of food waste (Almy, 1925). Under anaerobic conditions, waste containing organic sulfur compounds (cysteine and methionine) can be decomposed through the desulfurization process (Peu et al., 2012; Schieder et al., 2003; Chen et al., 2011). During the process, sulfur is desulfurized from organic molecules to

generate H_2S gas, as shown in Equation 7.

$$Organic - S \overset{Anaerobic}{\Rightarrow} H_2S \tag{7}$$

The putrefaction can play a role in H_2S production in some conditions. However, it is generally believed that the putrefactive H_2S production rate is insignificant compared with inorganic sulfate reduction as discussed above (Li et al., 2008; He et al., 2011).

2.2 Source of Sulfur in Landfills

The main source of sulfur in a landfill includes gypsum drywell (or wallboard), sulfur-containing organic waste (mainly food waste and paper), and sludge of wastewater treatment plants (WWTPs). One of the largest sulfur sources in landfills is gypsum drywall, especially in C&D debris landfills. The utilization of gypsum drywall by SRB to produce H_2S in landfills has been studied (Xu et al., 2010b; Fairweather and Barlaz, 1998). Gypsum is a mineral composed of calcium sulfate ($CaSO_4$) and water (H_2O; approximately 90% $CaSO_4 \cdot 2H_2O$ and 10% paper) and is widely used in commercial and residential construction and renovation for interior walls because it is a good fire barrier. Although waste gypsum drywall can be recycled, it often ends up in landfills (Musson et al., 2008).

Food waste is one of the main components of MSW (Fang et al., 2012). Some food waste, such as dairy products and meat products, contain organic sulfur compounds. Currently, the most common method of food waste treatment is landfilling, especially in developing countries (Fang et al., 2012). Typical sulfur content of food waste is known to be as much as 0.4% (by mass) by ultimate analysis (measured as C, H, N, O, S, and ash; Tchobanoglous et al., 1993). Under anaerobic conditions, organic sulfur compounds, such as cysteine and methionine, are biologically decomposed to produce H_2S (Ryckebosch et al., 2011; Wu, Wang et al., 2010; Wang et al., 2001). Paper waste also contains sulfate as measured up to a few tens of millimoles per kilogram of waste paper (Gurijala and Suflita, 1993). Watanabe et al. (2004) measured the sulfur of waste paper ranging from 0.40 to 1.7 mg/g (dry weight of waste paper) in Osaka, Japan. If landfilled MSW contains a large portion of waste paper, the waste paper can be considered an important sulfur source in landfills.

Wastewater sludge is a by-product generated in a WWTP. It is common practice to codispose sludge with MSW in landfills (Fairweather and Barlaz, 1998; Fang, Yang, Cen et al., 2012; Çinar et al., 2004). The sulfur content of wastewater sludge ranges from very little to 2.3% on a dry solid basis (Dewil et al., 2009). As sludge is generally dewatered prior to being landfilled, some chemical conditioners are often added to enhance sludge dewaterability. The use of the conditioners can affect H_2S generation in landfills. Liu et al.

(2012) conducted research to investigate the odorous gas compounds emission from different sludge conditioning processes. The results indicated that H_2S generation decreased with the addition of CaO as conditioner. However, the addition of H_2SO_4 and $FeSO_4$ for pH adjustment and Fenton reagent not only increased the sulfur content in sludge, but also accelerated the decomposition of sulfur-containing proteins (Liu et al., 2012).

2.3 Factors Affecting the Biological Formation of H_2S

Because H_2S generation mainly is a result of biological processes, its generation is firmly related to the growth of SRB. D'Imporzano et al. (2008) reported there was a good regression between biological activity and the odor molecules ($R^2 = 0.991$) during biodegradation of organic matters. The activity of SRB in landfills depends on many different factors, such as moisture content, temperature, and pH (D'Imporzano et al., 2008).

Moisture is essential for SRB growth. Bergersen and Haarstad (2008) observed that H_2S production significantly increased when waste is wet and water-saturated. It was observed that after a rainfall event the ambient H_2S concentrations were generally higher than those under normal conditions in landfills. The infiltration of rainwater into dry wastes can stimulate the activity of SRB resulting in H_2S generation (Johnson, 1986).

Temperature is another important factor for SRB growth. Sulfate reduction rates typically increase 2- to 3.9-fold with a temperature increase of $10°C$ (Widdle, 1988). The optimum temperature for most pure cultures of SRB ranges from $28°C$ to $32°C$. The lower optima among SRB are observed with some *Desulfobacterium* strains and a curved *Desulfobacter* strain at $24–28°C$, and the highest is around $70°C$ for *Thermodesulfobacterium commune* (Hao, 2003). Hao et al. (1996) reported that most species of SRB died rapidly at temperatures above $45°C$. Xu et al. (2010b) observed that in a field study, H_2S generation underneath a compost layer was reduced to some extent, due to high temperature (about $51°C$) in the layer. H_2S production from a landfill varies with seasons likely because of seasonal temperature variation. Kim (2006) measured seasonal variation of H_2S emission from two landfill sites in South Korea. The results indicated that H_2S had the highest emission rate during summer (Kim, 2006). Vasarevicius (2011) also observed that H_2S concentrations in the air near a landfill were related with seasonal changes of ambient temperature, because the generation of H_2S in the top layer of waste was affected by air temperature (Vasarevicius, 2011).

SRB prefer a neutral pH environment and are usually inhibited at pH values lower than 5.5 or higher than 9 (Hao et al., 1996). Research demonstrated that SRB can survive at low pH conditions (less than 4.5) like peat bogs and acid mine water. It was hypothesized that in acidic environments, SRB were present in microniches. In addition, the effect of low pH could be minimized because the sulfate reduction process generates additional bicarbonate alkalinity (Hao, 2003). Changing the pH value by adding acid or

alkali was reported as a method of inhibiting SRB in industrial plants (Postgate, 1984). Since pH of waste may vary with codisposed wastes, it has been recommended that prior to disposing gypsum drywall in landfills, powdered lime could be added to mix with gypsum drywall to reduce H_2S generation (Chalvatzaki and Lazaridis, 2010; Xu et al., 2011).

3. H_2S IN LANDFILL ENVIRONMENT

3.1   Levels of H_2S in Landfill Environments

H_2S measurements have been reported as concentrations in air, in landfill gas samples or flux rates from landfill surfaces. However, there is no standard method to measure H_2S emission from landfills. Various sampling methods have been used to collect landfill gas samples in previous studies. Gas samples often are collected from landfill gas collection systems (active or passive venting systems), the inside of landfills using soil gas probes installed though the landfill cover, ambient air near or over the site, and/or a chamber flux meter to capture H_2S emitted from cover soil (Lee et al., 2006; Xu et al., 2010b; Eun et al., 2007). H_2S measured with a single sampling technique probably is insufficient to describe the entire H_2S emission from a landfill because the levels of H_2S measured vary with different sampling techniques and sampling points. H_2S concentrations in gas collection systems can represent a relative amount of H_2S generation but does not include fugitive emissions because the gas in the collection system usually is treated before being released. Also, the forced extraction of landfill gas can be diluted landfill gas with ambient air that moves through the landfill cover by suction pressure. H_2S measurement using soil gas probes and flux chambers covers only a small portion of a landfill surface. The measured levels may not be representative for the whole area of the landfill surface. Direct measurements of H_2S from ambient air are easily influenced by meteorological conditions (temperature, atmospheric pressure, and precipitation). To properly measure H_2S emissions from a landfill, multiple measurement techniques may be required to understand landfill conditions and metrological parameters.

3.1.1 H_2S Concentrations in Landfill Gas

As summarized in Table 1, H_2S concentrations in landfill gas samples often range from under detection limits to thousands part per million. The UK Environment Agency (2002) reviewed trace landfill gas data from 79 sites. In the reviewed data, H_2S concentration was reported as high as about 70,000 ppm. This magnitude of H_2S concentration is not often observed in MSW landfill gases. The median concentration and average concentration of H_2S was 2.0 ppm (2.8 mg/m^3) and 96.6 ppm (134.2 mg/m^3), respectively (Parker et al., 2002). However, 68% of H_2S concentrations reviewed were under its detection limit. USEPA (1995) conducted an extensive measurement of landfill gases emitted from the Fresh Kills landfill at New York. The

TABLE 1. Health effects of hydrogen sulfide (Agency for Toxic Substances and Disease Registry, 2006)

Exposure limits (ppm)	Health effects
0.008–0.2	Olfactory threshold –"rotten eggs" smell detectable
20	Sense of smell to gas lost
	Concentrations tolerated for some hours without harm
20–50	Eye irritation
50	Prolonged exposure may cause pharyngitis and bronchitis
60	Prolonged exposure may cause conjunctivitis and eye pain
150	Irritation of upper respiratory tract; sense of smell lost
250	Pulmonary edema with risk of death
500	Very dangerous, evacuation should occur well below this level
1000	Loss of consciousness occurs
1000–2000	Immediate collapse with paralysis of respiration

concentrations of H_2S measured from passive vents at the Fresh Kills landfill ranged from 0.11 to 220 ppm. The ratio of H_2S production rate to methane production from the landfill was estimated to be 2.53×10^{-5} (by mass production rate). The emissions were measured from landfill surface and passive vents and the landfill gas collection system.

3.1.2 H_2S Emission from Landfills

Few field studies have been performed for H_2S emission. In the U.S. Environmental Protection Agency (USEPA) Fresh Kills landfill study, landfill gas samples were collected from four sections (two closed sections and two active sections) of the landfill using passive vent wells, flux chamber meters, and gas collection system (U. S. Environmental Protection Agency, 1995). Total emission rate for the Fresh Kills landfill was estimated at 0.453 g/s (emissions from landfill surface and passive vents). The observed surface emission ranged from below detection limit (< 0.29 mg/m²-day) to 10,498 mg/m²-day. The H_2S emission factor based on flux chamber measurement was estimated to be 4.34×10^{-10} g/s-kg of MSW. The magnitude of H_2S emission from C&D waste landfill is greater than in MSW landfill if C&D waste contains a high degree of sulfur-containing waste, like gypsum drywall. Eun et al. (2007) observed that H_2S emission rate was proportional to the gypsum drywall content. From measurements of five C&D landfills, they reported that the inverse distance weighting (IDW) mean emission rate ranged from 0.192 (± 0.34) to 1.76 (± 4.15) mg/m²-day. Colledge (2008) presented continuous monitoring data of H_2S from a C&D landfill site in Ohio, USA. The researcher estimated flux rates using Box model and data measured in the landfill to range from 0.005 mg/m²-day to 449.28 mg/m²-day.

3.1.3 Level of H_2S concentration in ambient air around landfills

H_2S is ubiquitous in ambient air, and concentrations from natural sources range between 0.11 and 0.33 ppb. Typical H_2S background concentrations

are known as less than 1 ppb (Agency for Toxic Substances and Disease Registry, 2006). In literature, H_2S concentrations in ambient air samples collected from landfill areas are much greater than the background level range. Table 1 shows ranges of H_2S concentrations from under detection limits to low 10s ppb. Even though there are some incidences of H_2S exposure with offensive levels, the exposure may not be chronic due to meteorological conditions strongly influencing the level of H_2S in the ambient air near an H_2S source.

3.2   Environmental Factors Influencing H_2S Emission

H_2S emission from a landfill is affected by various environmental factors, such as landfill geometry, waste characteristics (age and composition), the presence of air in the landfill (landfill aeration), and weather conditions. Geometrical H_2S emission from landfills can be related to waste landfill geometry such as area fill, trench fill, and valley fill (canyon fill). The preliminary H_2S production areas in an MSW landfill are an active waste tipping area. Barry et al. (2003) measured landfill surface flux emissions for 21 active landfills in the United Kingdom and found that the emission from landfill sides was greater than the top sides. In contrast, a USEPA report indicated that the average emission factor determined by flux chamber measurements on the side was similar to that on the top, but the total emission rate from the top was greater than that from the side because the surface of the top area was larger than that of the side (U. S. Environmental Protection Agency, 1995). However, the source of H_2S emission is not only landfilled waste itself but also other sources such as waste process area and leachate ponds in a landfill site. Fang et al. (2012) measured H_2S concentration in ambient air samples collected from eight locations in a landfill site accepting MSW and sewage sludge. They also collected gas samples from the gas extraction well of a closed landfill area. High levels of H_2S were measured from the active landfill tipping area (109 ppb), sludge discharging area (48.9 ppb), and leachate storage pool (53.0 ppb).

Waste age is a critical factor for H_2S generation. H_2S levels in gas samples from fresh waste materials are higher than those of old waste. Kim (2006) observed that the H_2S concentrations ranged from 0.4 to 524 ppm in gas samples collected vent pipes of a young landfill (<5 years old, lined landfill) but from 0.23 to 10.2 ppb in an old landfill (5–23-year-old, unlined landfill). In a UKEPA study, the highest concentrations of H_2S were also reported in the early stages of refuse compositions, and the levels of H_2S concentration gradually dropped over time (Parker et al., 2002). This observation may be due to the depletion of sulfur compounds over time by reductive conversions.

The presence of oxygen inside a landfill can influence the formation of H_2S in the landfill. Operating a landfill in aerobic conditions (adding air into

the landfill) is not common. However, aerating waste has been used to abate odor problems during remediation activities (Jacobs et al., 2003). Because hydrogen sulfide is produced under reduced conditions, the aeration creates unfavorable conditions for sulfur reducing bacteria and oxidizes H_2S. Powell et al. (2006) measured hydrogen sulfide reduction in a bioreactor landfill by air injection. The authors listed the causes of the reduction as a change in the reduced conditions by aeration and dilution and decomposition of H_2S by added air.

Meteorological conditions (temperature, atmospheric pressure, wind, and precipitation) influence landfill surface gas emission, H_2S dispersion in the air, and the formation of H_2S in a landfill (McBain et al., 2005; Poulsen et al., 2003; Christophersen et al., 2001). The sensitivity of gas emission from a landfill with change of atmospheric pressure has been studied. Typically, gas emission rates show negative correlation to atmospheric pressure change. Rain events can provide moisture SRB in a landfill, eventually. The moisture content of landfill soil cover also is a key factor of the gas emission. Landfill gas emission decreases with increasing cover soil moisture content (Christophersen et al., 2001). Due to high water solubility of H_2S, when H_2S gas passes through wet cover soil, it would tend to dissolve in water, temporarily reducing H_2S emissions. As water evaporates in cover soils, the accumulated H_2S gas is emitted, resulting in higher ambient H_2S concentration (Panza and Belgiorno, 2010). The impact of temperature on SRB was described in the previous section. Higher H_2S concentrations of the ambient air samples inside landfill footprints are reported during summer than those during fall.

4. IMPACTS ON THE ENVIRONMENT AND HUMAN HEALTH

4.1 Environmental Impact

4.1.1 FORMATION OF AIR POLLUTIONS

The emission of H_2S from landfills can result in the formation of another air pollutant, sulfur dioxide (SO_2). The released H_2S can react with atomic oxygen (O), oxygen (O_2), or ozone (O_3) to form SO_2, as shown by Equation 8 (Bibbero and Young, 1974). SO_2 also has a pungent odor, but its odor threshold (1 ppm) is higher than H_2S (Agency for Toxic Substances and Disease Registry, 2006). In general, the lifetime of H_2S before conversion to SO_2 is on the order of hours (Seinfeld, 1975). SO_2 is eventually removed from the atmosphere through absorption by plants and soils or through precipitation.

$$H_2S + O_3 \rightarrow H_2O + SO_2 \qquad (8)$$
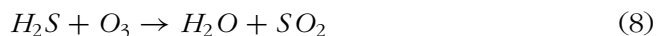
In addition, H_2S contained in landfill gas can also cause equipment damage (e.g., corrosion) in the facilities using landfill gas as an energy
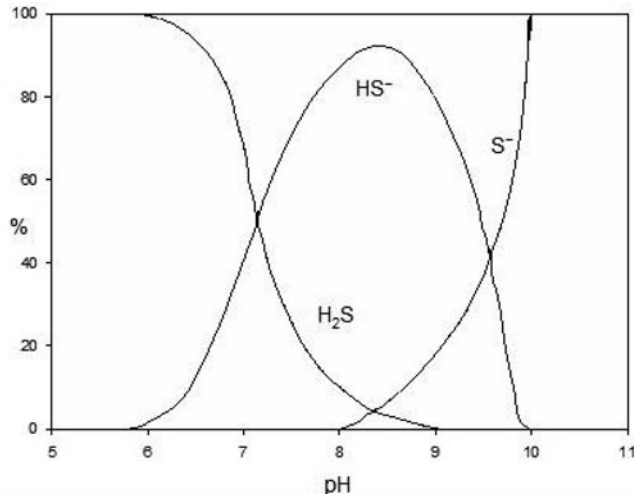
FIGURE 1. Ionic species of hydrogen sulfide (Hao et al., 1996).

source. It was reported that damage caused by H_2S scale is serious in many facilities. Therefore, pretreatment processes are required to remove any H_2S before landfill gas is used to produce heat and electricity (Nam et al., 2011).

4.1.2 GROUNDWATER CONTAMINATION

Compared to other chemical species in landfill gas, H_2S has relatively high water solubility (about 4,370 mg/L at 0°C, and 1,860 mg/L at 40°C). A relatively large amount of H_2S can dissolve into landfill leachate, an important reservoir of H_2S. Bergersen and Haarstad (2008) reported that leachate from Norwegian landfills emitted H_2S with concentrations as high as 5,000 ppm. The existence of ionic species of H_2S in water is highly dependent on pH. Major sulfur species present in water are H_2S, bisulfide (HS^-) and sulfide (S^{2-}). Changes in the concentrations of H_2S, HS^-, and S^{2-} are highly affected by pH changes. As shown in Figure 1, H_2S is the dominant species at a low pH between 5.0 and 6.0. At a pH of 7.0, the ratio of the concentration of aqueous H_2S to HS^- ion is approximately 1. When the pH is increased between 7.0 and 9.0, the HS^- becomes the dominant species. At a high pH greater than 9.0, the S^{2-} becomes the dominant species (Thompson et al., 1995).

The generation of H_2S from landfills can cause a serious threat to groundwater resources by two potential pathways to groundwater. Dissolved H_2S can migrate downward with leachate into groundwater, especially in old landfills without bottom liner systems. Otherwise, H_2S in landfill gas can travel through the unsaturated zone and come into contact with groundwater

(ElFadel et al., 1997). Groundwater contaminated by H_2S has an undesirable taste with a rotten egg smell. Ingestion of H_2S contaminated water can result in stomach discomfort, nausea, and vomiting (Thompson et al., 1995; Edwards et al., 2011).

4.2   Human Health Impacts

In addition to the odor problem, H_2S poses adverse impacts on human health (Flynn, 1998; Theakston, 2000; Selene and Chou, 2003; Campagna et al., 2003). The health hazard depends upon H_2S exposure duration, frequency, and concentration. Exposure to low concentrations of H_2S can cause difficulty in breathing for some asthmatics and irritation to the eyes, nose, and throat (Kansas Department of Health & Environment, 2010). Eye irritation has been described as the first health effect to manifest at low concentrations. In community settings, following short-term exposure, 25 ppb H_2S appears to be the lowest concentration observed to irritate the eyes. Serious eye damage is caused by a concentration of 50 ppm (70 mg/m^3; Lambert et al., 2006). As H_2S concentrations increase beyond 100 ppm, they quickly paralyze the olfactory senses so that the odor can no longer be recognized as a warning signal and it begins to affect the whole body. Exposure to high concentrations results in depression of the central nervous system and loss of consciousness. Over 500 ppm, H_2S can cause convulsions, respiratory arrest, coma, and even death (Flynn, 1998). Table 2   shows the health effects of respiratory exposure of H_2S at various concentrations.

The health and quality of life in landfill-neighbored communities and landfill workers has been investigated. Heaney et al. (2011) observed that there was a positive association of low-level H_2S exposure from a landfill with reports of malodor when the wind was blowing from   the landfill toward the community. The Agency for Toxic Substances and Disease Registry in the United States conducted an exposure   investigation of odor complaints from residents living within about a 2.4 km radius of the APAC-Reno landfill. The reported health problems associated with odors included difficulty   breathing, asthma exacerbations, headaches, and nausea. Dzaman et al. (2009) investigated the functioning of the taste and smell senses in workers employed at a landfill. The results showed that after working hours, the percentage of workers with smell disorders increased from 21.7% to 38.4% in the group of landfill workers because of the exposure to odor gases in the landfill. Fielder et al. (2000) compared the health conditions of residents living near the Nant-y-Gwyddon landfill site to a population with similar socioeconomic status. The monitoring data demonstrated that the site was responsible for the odor problem, and there was an increased maternal risk of having a baby with a congenital abnormality in residents near the landfill.

TABLE 2. Summary of H₂S levels measured from landfill sites

Site, Region	Waste composition/Landfill type	Season	Sample type	Sampling location/description	H₂S concentration/Emission	Reference
A landfill near Shanghai, China	MSW (7,600 tons/day) and sewage sludge (2,500 tons/day).	May 12 (spring), 2011 21°C, 55–65% RH, 5.7–6.9 m/s wind speed	Ambient air	TU: active landfill tipping area-U TD: active landfill tipping area-D LS: leachate storage pool LD: leachate disposal pool SD1: sludge discharge area SD2: sludge disposal work place SD3: sludge sun drying site SL: sludge landfill site	TU: 109 ppb TD: 3.9 ppb LS: 53.0 ppb LD: 7.1 ppb SD1: 6.6 ppb SD2: 48.9 ppb SD3: 3.3 ppb SL 2.9 ppb	(Fang et al., 2012)
Jerubaiciai landfill, Plunge district in Lithuania	MSW, 14900 m³ (2.5 thick) with 20–25 cm soil cover	August, November, February, and April	Ambient air	Center of the site, 1m above surface	900 ppb (February) 8600 ppb (August)	(Vasarevicius, 2011)
Five landfills in Beijing, China	MSW landfill	April	Ambient air	Chinese National Criteria, GB3095-1996) and Manual Methods for Ambient Air quality Monitoring (China Environmental Bureau HJ/T194-2005)	< 3.6 ppb	(Li et al., 2008)

Location	Waste	Sampling medium	Sampling period	Sampling locations	Concentrations	Reference
Daegu landfill, S. Korea	MSW Daegu Bangchun	Ambient air	January, 2004 Air temperature: −1.2–1.8°C	Landfill boundary and a near residential area These eight locations were representing (1) a residential site that is approximately 1km away, (2) the entrance point of the landfill, (3–7) five points evenly distributed on the landfill borderline, and (8) one in the central landfill area (covered area).	1: 0.001–2.1 ppb 2: 0.026–0.76 ppb 3: 0.25–1.19 ppb 4: 0.78–2.33 ppb 5: 0.61–7.57 ppb 6: 1.50–11.29 ppb 7: 0.43–14.49 ppb 8: 5.46–27.01 ppb	(Shon et al.,
Two landfills in S. Korea	Gunsan landfill (G-LF) Household waste Donghae landfill (D-LF) House hold waste + 20% coal ash	Ambient air (1 m above the ground)	G-LF: May (P1), August (P2), and December (P3) D-LF: May (P1'), July(P2'), October (P3'), and December (P4')	G-LF: G1: The central location, G2: a residential area about 2km away from the LF, G3: a suburban residential area about 1km away from the LF, and G4: a coastal area G4within 3 km of the LF. D-LF: D1: The central location of the LF (i.e., D1), D2-D4: the 3 sites are all within 3 km east of the LF.	G1: < 0.052–1.68 ppb G2: < 0.052–0.297 ppb G3: < 0.052–0.109 ppb G4: < 0.052–0.168 ppb D1:0.0 46–5.396 ppb D2: 0.023–0.101 ppb D3: 0.015–0.135 ppb D4: 0.016–0.194 ppb	(Song et al.,
Florida, USA	10 C&D landfills	Ambient air		Directly measured from ambient air using an analyzer	Average 3–4,000 ppb (Lee et al., 2006)	

TABLE 2. Summary of H$_2$S levels measured from landfill sites *(Continued)*

Site, Region	Waste composition/Landfill type	Season	Sample type	Sampling location/description	H$_2$S concentration/Emission	Reference
Belgium	NA	NA	Ambient air in waste handling and discharging site	Sampling ambient air by pumping 20 dm^3 through adsorbent trap (sampling was performed at 1.5 m height)	Not detected	(Termonia and Termonia, 1999)
Tianziling landfill, Hangzhou City, China	Domestic waste (not accepted: medical, industrial, and hazardous wastes)	July, 2007–June, 2009 Air temperature: 5.2–28.5°C	Ambient air samples at 6 sampling points	Dumping area Wastewater treatment plant Office area Factory boundary Residential area (affected area by the odor) 165 platform (old dumping site)	369.1 ±16.8 ppb 350.3 ± 12.6 ppb 147.1 ± 6.6 ppb 128.0 ± 4.1 ppb 40.6 ± 2.0 ppb 129.1 ± 18.0 ppb	(Ying et al., 2012)
Warren recycling landfill, Ohio, USA.	C&D, the operation began in 1994.	Oct., 2003-Sep., 2004 June 9-August 15, 2004 Sep 8- Oct 6, 2004	Ambient air (continuous). Hourly averages (up to 7500 observation in 337 day period)	Two sampling stations (north [N] and northwest [NW]), sampling inlet located at the elevation of approximately 3.5 meters. Six additional sampling locations (Site 1 to 6) Data were normalized to STP (25°C and 29.92"Hg)	N: 0–87 ppb NW: 0–178 ppb Site 1: 0–141 ppb Site 2: 0–71 ppb Site 3: 0–15 ppb Site 4: 0–80 ppb Site 5: 0- 9 ppb Site 6: 0- 44 ppb	(Colledge, 2008)

Location	Type	Sampling period	Medium	Description	Results	Reference
MSW regional landfill, NC, USA.	MSW	Jan.- Feb, 2009 and Sep.-Nov. 2009	Ambient air	The community boundary near the landfill using continuous monitoring	Range: 0–14.86 ppb The 15-min average H_2S = 0.22± 0.52 ppb	(Heaney et al., 2011)
In the Valencianne Community, Spain		July, and November, 2009	Ambient air	Inside of landfill footprint Summer Autumn Outside of landfill Summer Autumn	3.6–16.3 ppb 2.5–8.8 ppb 0.2–4.3 ppb 0.01–2.9 ppb	(Font et al, 2010)
Jerubaiciai landfill, Plunge district in Lithuania	MSW	August, November, February, and April	Landfill gas	Two gas wells (G1 and G2)	August: 7.4 ppm (G1), 8.6 ppm (G2) November: 5.5 ppm (G1), 8.1 ppm (G2) February: 2.5 ppm (G1), 6.0 ppm (G2) April: 5.0 ppm (G1), 7.3 ppm (G2)	(Vasarevicius, 2011)

TABLE 2. Summary of H$_2$S levels measured from landfill sites *(Continued)*

Site, Region	Waste composition/Landfill type	Season	Sample type	Sampling location/description	H$_2$S concentration/Emission	Reference
A landfill near Shanghai, China	MSW (7,600 tons/day) and sewage sludge (2,500 tons/day).	May 12 (Spring), 2011. 21°C, 55–65% RH, 5.7–6.9 m/s wind speed	Two parallel samples were collected in each sampling point. All Tedlar bags were placed in a vacuum-generating container. Using a vacuum pump connected to the container, five liters of gas sample were drawn directly into the Tedlar bag via Teflon tubing with a 5 mm outside diameter inserted into the vent system. These samples were then transported immediately to the lab for analysis (within 24 h) to minimize the loss of reduced sulfur compounds due to storage.	GW: Closed landfill site, gas extraction wells	GW 16.1 ppm	(Fang et al., 2012)

Four MSW landfills in S. Korea	MSW	9 Aug.2003 22–23 July 2003 22–23 July 2003 13–16 Jan. 2004	Landfill gas	Ventpipe (from moderately fresh burial locations) W: Woon Jung landfill (active) N: No Hyung landfill (closed) H: Hoei Chun landfill (active) B: Bang Chun landfill (active)	0.212–681.370 ppm (W) 0.336 ppb ppm (N) 854.580–5,142.9 ppm (H) 89.132–143.091 ppm (B)	(Kim et al., 2005)
Dong Hae, S. Korea	MSW Young landfill (YLF) Old landfill (OLF)	May, July, November, and December	Landfill gas (venting pipe)	YLF: young <5 years, area: 0.35 km^2 OLF: old, 5–23 years, area: 0.32 km^2	YLF (ppm) OLF (ppm): May: 2.629–124.41 0.0002–0.002 July: 6.848–523.838 0.00055 Nov.: 0.437–280.97 0.0013–0.005 Dec.: 70.280–181.337 0.00063–0.01	(Kim, 2006)

TABLE 2. Summary of H$_2$S levels measured from landfill sites *(Continued)*

Site, Region	Waste composition/Landfill type	Season	Sample type	Sampling location/description	H$_2$S concentration/Emission	Reference
A landfill near Shanghai, China	MSW (7,600 tons/day) and sewage sludge (2,500 tons/day).	May 12 (Spring), 2011. 21°C, 55–65% RH, 5.7–6.9 m/s wind speed	Two parallel samples were collected in each sampling point. All Tedlar bags were placed in a vacuum-generating container. Using a vacuum pump connected to the container, five liters of gas sample were drawn directly into the Tedlar bag via Teflon tubing with a 5 mm outside diameter inserted into the vent system. These samples were then transported immediately to the lab for analysis (within 24 h) to minimize the loss of reduced sulfur compounds due to storage.	GW: Closed landfill site, gas extraction wells	GW 16.1 ppm	(Fang et al., 2012)

Site	Waste type	Date		Description	Values	Reference
Four MSW landfills in S. Korea	MSW	9 Aug.2003 22–23 July 2003 22–23 July 2003 13–16 Jan. 2004	Landfill gas	Ventpipe (from moderately fresh burial locations) W: Woon Jung landfill (active) N: No Hyung landfill (closed) H: Hoei Chun landfill (active) B: Bang Chun landfill (active)	0.212–681.370 ppm (W) 0.336 ppb ppm (N) 854.580–5,142.9 ppm (H) 89.132–143.091 ppm (B)	(Kim et al., 2005)
Dong Hae, S. Korea	MSW Young landfill (YLF) Old landfill (OLF)	May, July, November, and December	Landfill gas (venting pipe)	YLF: young <5 years, area: 0.35 km^2 OLF: old, 5–23 years, area: 0.32 km^2	YLF (ppm) OLF (ppm) May: 2.629–124.41 0.0002–0.002 July: 6.848–523.838 0.00055 Nov.: 0.437–280.97 0.0013–0.005 Dec.: 70.280–181.337 0.00063–0.01	(Kim, 2006)
Fresh Kills landfill, United States	Residential waste		Landfill gas (concentration [ppm] and emission rate ($\mu g/s$), surface emission.	Passive vents (H_2S concentrations, flow rates) Surface emission (flux chambers) Gas collection system (one active section out of four sections)	Passive Vents Section 2/8: 1.21–104 ppm Section 3/4: 0.11–220 ppm Section 1/9: 5.3–35 ppm	(Parker et al., 2002)
Sudokwon landfill, MSW Inchon city in S. Korea			Landfill gas	Extraction well	15.1–427.5 ppm	(Shin et al., 2002)

TABLE 2. Summary of H$_2$S levels measured from landfill sites *(Continued)*

Site, Region	Waste composition/Landfill type	Season	Sample type	Sampling location/description	H$_2$S concentration/Emission	Reference
Biovale Company landfill			Landfill gas	Landfill gas measured from landfill gas collection system	100± 10 ppm	(Ducom et al., 2009)
Florida, USA	10 C&D landfills		Landfill gas	Existing gas collection system or soil vapor probe (0.3 m depth)	< 0.003–12,000 ppm	(Lee et al., 2006)
Belgium	NA	NA	Landfill gas (bio-gas)	Sampled at a flare station	0.072 ±0.029 ppm	(Termonia and Termonia, 1999)
UK codisposal site	MSW (67%) and trade waste (23%) Young waste (3 years) Old waste (17 years)	September and March in 2001 and 2002.	Gas wells	Landfill gas	1.2–5.4 ppm 5.1–16.4 ppm	(Parker et al., 2002)
Five waste disposal sites at Mexico City		September-November, 2003	From drilled holes through the cover or From vent pipes/burners	Operating controlled dump Operating open dump Operating landfill Closed open dump Closed controlled/open dump	18 ± 5 ppm 50 ± 10 ppm 95 ± 10 ppm 120 ± 20 ppm 150 ± 20 ppm	(de la Rosa, Velasco, Rosas and Volke-Sepulveda, 2006)
Shanghai solid waste landfill, Shanghai, China		December, 2007, air temperature 14–18°C	Landfill gas, installing soil gas probes (80 cm depth) under the soil cover.	Cell 18 (< 1 year old) Cell 56 (2 years after the closure) Cell 52 (5 years after the closure) Cell 40 (13 years after the closure)	Cell 18: 84–3,400 ppm Cell 56: 0–67 ppm Cell 52: 0–4 ppm Cell 40: not detected	(Xiaoli et al., 2011)

Location	Site description	Sampling	Method		Results	Reference
Florida, USA	Site A: opened in 1991, 4% drywall; Site B: opened in 1990, 10% drywall; Site C: 35 m deep; 0.6-m final cover; Site D: opened early 1980s; Site E: closed in 1983	April-August	Flux chamber	Subsurface	Inverse distance weighted mean Site A: 0.19 ± 0.34 mg/m^2-day Site B, 1.76 ± 4.15 mg/m^2-day Site C, 1.53 ± 4.08 mg/m^2-day Site D, 1.47 ± 4.83 mg/m^2-day Site E, 0.54 ± 2.35 mg/m^2-day	(Eun et al., 2007)
Dong Hae, S. Korea	MSW Young landfill (YLF) Old landfill (OLF)	May, July, November, December	Total emission rate (estimated using H$_2$S concentrations measured in landfill gas from venting pipes)	YLF: young <5 years, area: 0.35 km^2 OLF: old, 5–23 years, area: 0.32 km^2	YLF: 232–278,211 g/year OLF: 0.03–3.72 g/year	(Kim, 2006)

TABLE 2. Summary of H$_2$S levels measured from landfill sites *(Continued)*

Site, Region	Waste composition/Landfill type	Season	Sample type	Sampling location/description	H$_2$S concentration/Emission	Reference
Fresh Kills landfill, NY, United States	MSW	Four sections: 3/4 section: closed (being retrofitted with: 1) passive vents, 2) PVC cover, and active gas collection wells. 2/8 section: closed (being retrofitted with: 1) passive vents, 2) PVC cover, and active gas collection wells.) 6/7 section: active: (covered with 0.3–0.6 m soil cover, no active gas collection, passive vents or PVC cover.) 1/9 section: active (active gas collection cover 2/3 of the section)	Landfill gas from passive vents or using flux chamber	Passive vents (emission rates were calculated with H$_2$S concentrations and flow rates) Surface emission (flux chambers)	Section 2/8: (n = 10) 0.0.30–3.99 mg/m²-day Section 3/4: (n = 11) 0–7.55 mg/m²-day Section 1/9: (n = 10) 0.19–10.94 mg/m²-day Section 6/7 (n = 43): 0.45–10497.6 mg/m²-day	(Parker et al., 2002)

Warren recycling landfill, Ohio, USA.	C&D, three phases. 70 acres for each phase. The operation began in 1994.	Oct. 17, 2003-Sep 17, 2004	Ambient air	AERMOD steady-state plume model was used to estimate downwind concentrations from landfill area sources. A simple box model (Box A-D) was used to back-calculate flux rates of on-site area sources. Continuous 5-minute ambient H_2S measurements were used.	Box A: 27.13 mg/m^2-day (Colledge, 2008) Box B: 29.72 mg/m^2-day Box C: 19.01 mg/m^2-day Box D: 210.82 mg/m^2-day

TABLE 3. Summary of methods for controlling H_2S in landfill gas

Purpose	Media	Category	Method	Examples
Inhibition of H_2S generation	Solid waste	Waste composition	Sulfur source removal/modifying waste composition	Sorting sulfur containing waste materials such as drywalls
	Solid waste	Biological treatment	Increasing pH (> pH9), increasing redox potential greater than −100 mV	NaOH, Ca(OH)$_2$, adding alkaline wastes like concrete
			Adding alternative electron acceptors	Air addition, nitrate addition
		Chemical treatment	Inhibition of SRB activities with chemical inhibitors	Sodium molybdate Ferric iron
Removal of H_2S formed	Landfill gas captured	Thermal treatment	Incineration	Flare stack
		Physicochemical treatment	Dry H_2S removal	Solids absorbents (iron oxides, zinc oxides, and alkaline solids) or adsorbents (zeolites and activated carbon)
			Liquid H_2S removal	Redox processes, alkaline solution processes, and amine solution processes
		Biological treatment	Biological oxidation H_2S	Bioscrubbers
				Biofilter Trickling biofilter Membrane bioreactors Activated sludge
	Landfill gas fugitive emission	Alternative cover materials	Biofilters Metal oxide filters	Compost
				Fly ash
		Masking agent	Deodorants or Neutralizer	Spray chemicals around or over odor source

## 5. H_2S EMISSION CONTROL TECHNOLOGIES

Suppressing H_2S generation and/or emission from a landfill can be achieved by inhibiting H_2S generation and removing H_2S from landfill gas. Table 3

summarizes different categories of H_2S reduction technologies. Options for H_2S control in landfills vary. Control techniques can be applied to remove the sulfur source from waste, to inhibit SRB growth in landfilled waste, and to remove H_2S from the landfill gas. H_2S removal in landfill gas methods also differs depending on the presence of a landfill gas collection system and the use of collected landfill gas.

5.1 Source Control

Controlling waste composition is one of the most effective methods for reducing H_2S generation from a landfill. The landfilling of biodegradable waste materials with high sulfate content has been prohibited in England and Wales since July 2005 (UK Health Protection Agency, 2011). Banning sulfur-containing waste such as drywall (usually 90% $CaSO_4 \cdot 2H_2O$ and 10% paper by mass composition) can inhibit SRB activities inside the landfill. Codisposal of C&D waste with high pH waste can reduce H_2S production. For example, waste such as concrete impacts H_2S generation by increasing pH greater than the optimum pH ranges (pH 6~9) for SRB and by absorbing H_2S that is present in landfill gas (Kenton et al., 2006).

H$_2$S formation in landfilled waste is influenced by codisposed materials. Yang et al. (2006) showed that H_2S generation is affected by the presence of codisposed wood and concrete. The organic acids leaching from codisposed wood lowered the pH of the leaching solution and deceased SRB activity. Concrete plays a role in the reduction of H_2S production by increasing leachate pH from optimal pH range of SRB and reacting with H_2S as the following chemical reaction (Equation 9). It was reported that mixing lime with gypsum drywall prior to disposing in landfills effectively reduced H_2S generation (Johnson, 1986; Xu et al., 2011).

$$\text{CaO} + H_2S \rightarrow CaS + H_2O \tag{9}$$
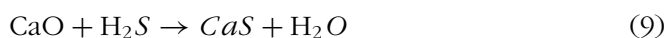
5.2 Use of Inhibitors for SRB

As H_2S from landfills mostly is produced biologically, creating artificial environments to inhibit SRB is an effective way to minimize H_2S emission. There are various ways to create unfavorable environments including changing redox potential, increasing pH, stimulating growth of a competing group of anaerobic bacteria, and adding SRB inhibitors.

The redox potential of the environment should be –100 mV or less to cultivate SRB (Postgate, 1984). The redox potential can be adjusted by adding oxidizing agents such as oxygen and nitrate. The inhibition of sulfate reduction could occur by increasing dissolved oxygen in wastewater greater than 1 mg/L (Hao et al., 1996). The addition of oxygen (air) not only increases

redox potential but also causes the abiotic oxidation of H_2S. Duangmanee (2009) studied sulfide removal from biogas of anaerobic digesters using precisely regulated air injection, referred to as microaeration, to both the gas and liquid phases (effluent). As a result of the investigation, more than 98% of sulfide in biogas was removed and 95% of the H_2S removal was caused by abiotic oxidation.

The addition of a chemical (an electron acceptor for a group of anaerobic bacteria outcompeting SRB) can result in inhibiting H_2S generation. The chemicals include nitrate and ferric compounds. The addition of nitrate into an aqueous environment also inhibits SRB by increasing the redox potential higher than –100 mV (Hao et al., 1996). In addition, it also stimulates the growth of nitrate-reducing bacteria (NRB), which have a large thermodynamic advantage over the SRB for the same substrates (Eckford and Fedorak, 2002). The addition of ferric iron can stimulate the growth of iron-reducing bacteria (IRB) which can outcompete SRB by maintaining concentrations of substrates at levels lower than thresholds required by SRB (Lovley, 1991). Lovley and Phillips (1986) found that the addition of ferric iron to sediment inhibited sulfate reduction by 86–100% (Lovley and Phillips, 1986). Du et al. (2014) reported that with the addition of $Fe(OH)_3$, the emission of H_2S decreased about 95% from biodegradable sulfur-containing waste, indicating iron has the potential role to control H_2S emission in landfills (Du et al., 2014). Xu et al. (2011) demonstrated the addition of ferric chloride (lowing pH) or hydrated lime (increasing pH) out of the optimum range for SRB can effectively inhibit H_2S generation. SRB is usually inhibited at pH values lower than 5.5 or higher than 9 (Hao et al., 1996).

Saleh et al. (1964) reviewed and summarized chemical inhibitors of SRB activities. The chemical inhibitors include antibiotics, detergents, dyes, mercurials, metal ions and complexes, nitro compounds, phenolic substances, sulfate analogs, and sulfonamides. Analogs of sulfate ion such as molybdate, chromate, tungstate, and selenate inhibit sulfate reduction. Among these metal ions, molybdate has been extensively used as an SRB inhibitor in lake and marine sediment and anaerobic digesters. Patidar and Tare (2005) showed that molybdate supplementation inhibited SRB cavity in biomass over 85% at 3mM. They found that the higher molybdate dose needed as the COD/SO_4^{2-} ratio increased could stimulate or inhibit the total methanogenic activity, depending on other factors. The concentration of formaldehyde (a range of 12–19 mg/L) effectively reduced sulfide generation up to 90% of the total sulfide with negligible impacts on COD removal, nitrification rate and oxygen uptake rates under anaerobic conditions (Zhang et al., 2008). However, the authors pointed out that the use of formaldehyde for an SRB inhibitor should be performed under caution because of the toxicity of the chemical to humans. Xu et al. (2011) observed considerable inhibitive effects of sodium molybdate (Na_2MoO_4) on H_2S generation from flasks with 100 g gypsum drywall incubated for 30 days (under 35°C). The average H_2S

concentration with 2 mM sodium molybdate dose was about 8.6 ppm, but the H_2S concentration in flasks without sodium molybdate showed up to 67,500 ppm (Xu et al., 2011). Even though the chemical inhibitors successfully reduced H_2S generation from simulated landfills, for the application of chemical inhibitors in landfills some issues should be addressed, including homogeneous contribution of inhibitors on waste, cost-effectiveness, and environmental impacts.

5.3 Removal of H_2S Formed in Landfill Gas

### 5.3.1 REMOVAL OF H_2S IN CAPTURED LANDFILL GAS

There are various methods for treating H_2S in landfill gas including thermal, physicochemical, and biological methods to treat H_2S in landfill gas. A typical modern sanitary landfill is required to be equipped with a landfill gas collection system and the final cover system. The most common option for collecting landfill gas is gas incineration on a candlestick flare system. This option results in converting H_2S to SO_2 or H_2SO_4.

If landfill gas is used for biogas to energy, it may be necessary to remove the impurities (including H_2S) of the gas below certain levels. Diverse H_2S removal processes for off-gases are developed and commercialized. Various processes to remove H_2S in biogas have been well reviewed by researchers (Jensen and Webb, 1995; Syed et al., 2006; Abatzoglou and Boivin, 2009). To select an appropriate H_2S removal technology for a landfill site, multiple variables such as concentration of H_2S gas in the bio gas; pressure, temperature and composition of the gas; ratio of H_2S to CO_2; volume of gas, total H_2S load, gas specifications for the process output, economic considerations, and environmental implications should be considered (Gómez and Cantero, 2007). In this section, common physicochemical and biological technologies to remove H_2S in biogases are summarized.

Physicochemical processes for removing sulfur compounds in gas (not only H_2S) include dry removal processes and liquid removal processes. In the dry processes, H_2S is removed by solid adsorbents. The dry absorbent processes are cost-effective with low sulfur loading but not economical for high sulfur loading because of spent waste production. Iron oxides are commonly used as H_2S adsorbents with an activator (0.125–5% weight basis; Abatzoglou and Boivin, 2009). Iron sponge (powdered iron oxide adhered to wood shavings) has long been used for the H_2S removal from the gas phase. To remove H_2S, gas is forced to a container or a series of containers filled with iron sponge. The iron oxide of iron sponge reacts with H_2S to form iron sulfide. The used iron sponge can be regenerated back to iron oxide by exposing the bed to oxygen. Xu et al. (2010a) used oxidized tire-derived steel, a by-product of automobile tire recycling, as alternative cover material for H_2S removal. The oxidized steel showed a much stronger H_2S
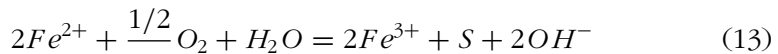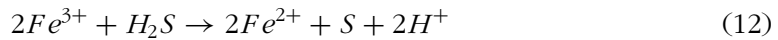
removal ability than sandy soils. With an initial concentration of 550 ppm, over 98% H_2S was removed in 2 min by 20 g of the steel, compared to only 50% H_2S removal in 60 min by the same amount of sandy soil (Xu et al., 2010a). Chemical reaction involved in iron oxide adsorbents are as shown in Equation 10 and Equation 11.

$$Fe_2O_3 + 3H_2S \rightarrow Fe_2S_3 + 3H_2O \qquad\qquad (10)$$

$$Fe_2S_3 + 3/2O_2 \rightarrow Fe_2O_3 + 3S \qquad\qquad (11)$$
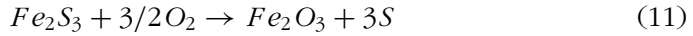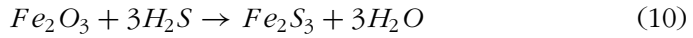
Commercially available iron oxide absorbents include SulfaTreat (MI SWACO), Sulfur-Rite (Merichem), Media-G2 (ADI International Inc.), and GAS RAP. Few cases of the use for treating landfill gas were reported (UK Environment Agency, 2010). Abtzpglou and Boivin (2009) compared the performances and costs of these iron sponges.

Impregnated carbon is also widely used in gaseous H_2S removal processes, which combines both physical adsorption and chemical reaction. Carbon source, such as coconut shell, serves as an adsorbent for H_2S, while the chemicals, such as metal oxide and alkaline materials, react with H_2S. Other absorbents, such as silica gel, alumina, zeolites, etc., can also be used to remove H_2S in landfill gas (Xiao et al., 2008; Elsayed et al., 2009).
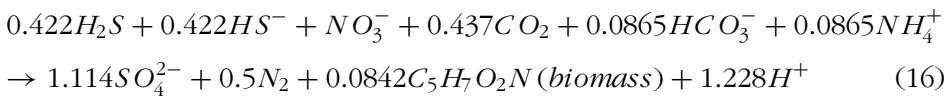
Liquid H_2S removal processes include liquid redox processes, alkaline solution processes, and amine solution processes. In liquid redox processes, iron-chelated solutions convert sulfide to element sulfur by the reactions expressed as shown in Equations 12 and 13.

$$2Fe^{3+} + H_2S \rightarrow 2Fe^{2+} + S + 2H^+ \qquad\qquad (12)$$

$$2Fe^{2+} + \frac{1/2}{} O_2 + H_2O = 2Fe^{3+} + S + 2OH^- \qquad\qquad (13)$$

LO-CAT (Shell/Dow) and Sulferox (US Filter/Merichem) processes are the commercial names of typical liquid redox processes. LO-CAT has been applied for treating H_2S in landfill gas in Florida, USA (UK Environment Agency, 2010). The economical range of the sulfur loading for LO-CAT and Sulferox is higher than 100–20,000 kg sulfur/day and more than 200 kg sulfur/day, respectively (Abatzoglou and Boivin, 2009). Alkaline solution processes are not economically appropriate for landfill gas because this process produces a large volume of waste stream, which is nonregenerable by the reaction of alkaline salts and CO_2 of the landfill gas. Amine solution processes are used in applications processing very large quantities of sulfur in gas (>27,000 kg/day), such as petroleum refinery processes (Baspinar et al., 2011). Other liquid-based techniques applied to reduce H_2S in landfill gas include water scrubbing (Sonzy landfill, France) and the use of Selexol (liquid solvent; UK Environment Agency, 2010).

Estrada et al. (2011) compared different odor-treatment technologies for wastewater treatment plants in terms of environmental performance, process economics, and social impact using the IChemE Sustainability Metrics. In their study, biological treatment technologies showed the highest investments but the lowest operating cost. However, the investment costs per unit flow rate treated decreased dramatically when the design flow rates increased. Meanwhile, physical/chemical technologies exhibit higher environmental impacts than biological technologies in terms of energy consumption, material and reagents consumption, operational risk, and hazardous waste production (Estrada et al., 2011).

Biological oxidation of sulfide into elemental sulfur can be conducted by chemoautotropic or photoautotropic bacteria, as shown in Equations 14 and 15. Also, oxidation of sulfide can occur in the absence of oxygen with nitrate (Equation 16; Tichý et al., 1998).

$$2H_2S + O_2 \rightarrow 2S^0 + 2H_2O \text{ chemoautotrophic process} \tag{14}$$

$$2H_2S + CO_2 + hv \rightarrow 2S^0 + CH_2O + H_2O \text{ photoautotrophic process} \tag{15}$$

$$0.422H_2S + 0.422HS^- + NO_3^- + 0.437CO_2 + 0.0865HCO_3^- + 0.0865NH_4^+$$

$$\rightarrow 1.114SO_4^{2-} + 0.5N_2 + 0.0842C_5H_7O_2N\,(biomass) + 1.228H^+ \tag{16}$$
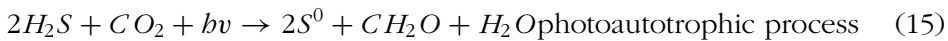
Biological H_2S removal processes with various microorganisms are well documented by Jensen and Webb (1995) and Syed et al. (2006). Syed et al. (2003) suggested that the use of phototrophic bacteria (e.g., *Cholorobium limicola*) for the removal of H_2S in biogas was beneficial for the anaerobic nature of the gas, even though light supply is a key constraint of phototrophic oxidation of H_2S. However, chemoautotrophic thiobacteria more often are used to treat H_2S in biogas. Among the chemoautotrophic oxidation bacteria, *Thiobacillus* sp. are commonly used in H_2S removal studies. *Thiobacillus* sp. have the ability to grow under various environmental stress conditions such as oxygen deficiency, acid conditions, and etc. The common biological treatment processes for gases purification include biofiltration, bioscrubbing, and biotrickling filtration (Zdeb and Pawlowska, 2009).

5.3.2 H_2S Removal in Fugitive Landfill Gas

5.3.2.1 Use of Alternative Covers. Landfill cover is an essential component for controlling gas emission from a landfill. Contrasted with conventional cover soil (earth material), various alternative cover materials have been proposed to abate odor emission. Suggested alternative cover materials by previous studies include compost, wood chips, fine waste materials (construction and demolition fines, commercial and industrial fines, and crushed concrete), fly ash, bottom ash (wood ash, coal ash, and municipal waste incinerator bottom ash), quicklime, hydrated lime, calcium carbonated materials, and waste steel (Plaza et al., 2007; Xu et al., 2010a; Xu et al., 2010b;

Solan et al., 2010). Each alternative cover material was tested independently or by mixing with other material. Physical adsorption, chemical reactions, and biological oxidation are the main mechanisms of H_2S reduction of alternative landfill cover materials. However, the capacity of H_2S removal with alternative cover materials can vary with experimental conditions and physicochemical properties of the materials.

Biofilter materials (e.g., wood chips, yard waste, compost) are often available in landfill sites that process these wastes. Hurst et al. (2005) examined the ability of municipal waste compost as a daily cover material for reducing odorous emission. In their study, the level of H_2S (70 ppm) in simulated odor gas considerably decreased by more than 95% with the first 10 cm of compost (with a bulk density 590 kg/m^3). Bergersen and Haarstad (2008) compared various filter materials (spruce bark, wood chips, sludge compost, the mixture of bottom ash to iron oxide [3:1], and iron oxide) for H_2S removal in the gas generated from simulated columns with different water levels. They concluded that metallic filter materials had greater adsorption potentials than those of organic filter materials. The sulfide adsorption of different alternative cover materials for odor reduction is summarized in Table 4.

MSW incineration bottom ash has been shown to be useful as sorbing agent for remove H_2S from landfill gas (Mostbauer et al., 2008). Ducom et al. (2009) observed that 1 kg of MSW incinerator bottom ash could remove more than 3.0 g of hydrogen sulfide. H_2S retention capacity of bottom ash relied on moisture content and contact time (gas flow rates) in their experiments. They also explained the retention mechanisms of H_2S by bottom ash as acid-basic reactions (sulfide ion [S^{2-}] in the basic medium reactions with metal cations in solution). He et al. (2011) tested adsorption removal of H_2S with waste biocover soil collected from a rural waste bioreactor and sieved to separate large pieces, with different particle sizes (≤ 0.45 mm to 5 mm), temperature (4–45°C), moisture (2–60%), pH (3–11), and oxygen content (0–21%). They observed that the absorption capacity of the biocover increased with decreasing particle size, increasing moisture content, and increasing pH. Also, they found that H_2S removal varied with different oxygen contents. Peak absorption capacities were observed at 35% moisture content and 10% oxygen content in the tested ranges. The influence of oxygen on H_2S removal with alternative cover materials was observed in other research (Xu et al., 2010a). By the adsorption isotherm, the authors concluded that H_2S concentration, the characteristic of water film on adsorbents (pH, temperature and ionic strength), and the characteristics of absorbents (surface area, organic matter, mineral composition, and pore size) played important roles in H_2S adsorption characteristics.

The amendment of cover soil also is another option to reduce H_2S emission. Mixing on-site soil with alkaline material(s) such as quicklime, hydrated lime, and/or fine concrete materials, has been tested in the laboratory and

TABLE 4. H_2S removal using alternative cover materials

Materials	Sulfide adsorption (mg/kg) or H_2S removal	Experimental conditions	Reference

Spruce bark
  Wood chips
  Sludge compost
  Bottom ash/iron oxide 3:1
  Iron oxide
	215 mg S/kg
	387 mg S/kg
	762 mg S/kg
	3,345 mg S/kg
	
	983 mg S/kg
		Column tests by adding a filter layer (5–10 cm)	(Bergersen and Haarstad, 2008)

Waste biocover soil	3–56 mg/kg (particle size <0.45 to >25 mm)
	8–168 mg/kg (pH 3–11)
	34–60 mg/kg (O_2 0–10%)
	383 ± 40 mg/kg (saturation)
		Batch experiment
		  Particle size, <0.45 to >25 mm
		  Water content: 2–60%, pH 3–11, O2 0–10%
			(He et al., 2011)

Coal bottom ash pH 6..2–8.6, surface area (m^2/g) 7.2–15.5	10.5 mg H_2S/g bottom ash	Gas adsorption column. Gaseous H_2S (9000 ppm) mixed with pre-humidified air	(Lin et al., 2001)
MSW incineration bottom ash	3g H_2S/kg bottom ash	The experiment was designed for removing sulfur gas in landfill gas 90–150 mg Nm^{-3} H_2S (landfill gas) 13 kg bottom ash (MC 15%), flow rate 879–1710 $NLhr^{-1}$	(Ducom et al., 2009)

Red clay
  Compost
  Waste wire
  Sand
	36.04 H_2S mg/g
	25.97 H_2S mg/g
	29.82 H_2S mg/g
	17.39 H_2S mg/g
		Batch test. A 20 g sample with 5,090 ppm H_2S in 275 mL bottle (for 60 min)	(Liu et al., 2011)

Sandy soil Tire-derived steel	50% removal 99% removal (7.5 mg H_2S/g tire-derived steel)[†]	20 g sample in 550 ppm H_2S containing 275 mL bottle (tested for 60 min)	(Xu et al., 2010b)
MSW Compost with a density of 590 kg/m^3 and 740 kg/m^3 (moisture content 35–50%)	More than 90% H_2S and 95% H_2S in standard gas (70 ppm) was removed with the first 10 cm of 590 kg/m^3 and 740 kg/m^3, respectively. 1500–1600 mgH_2S/kg compost[†]	Column test. Odor reduction was measured in different depths (10-60cm) of compost layer using standard gas (a gas mixture of odor compounds) using a flow rate of 250 mL/min for 96 min	(Hurst et al.,

[†]Calculated adsorption capacity with given information but may not be at equilibrium conditions.

field (Plaza et al., 2007; Xu et al., 2010b). Xu et al. (2010b) found that fine concrete, compost, sandy soils mixed with 10% $CaCO_3$, 1% $Ca(OH)_2$, or 3% $Ca(OH)_2$ effectively reduced H_2S emission from a C&D landfill in their field and laboratory tests. These alkaline materials not only remove H_2S in landfill gas by the reactions to form sulfide minerals but also possibly reduce H_2S generation by creating alkaline environments under the amended cover materials.

Some organic filter waste from production of mineral wool also showed good H_2S removal capacity (Bergersen and Haarstad, 2008). Bergersen and Haarstad (2014) investigated the efficiency of H_2S removal from landfill gas using metal rich mineral wool wastes. Results show that the H_2S gas can effectively be removed by mineral wool waste and the estimated sulfide removal potential was 14 g/kg for mineral wool and 261 g/kg for rod mill waste, respectively (Bergersen and Haarstad, 2014).

5.3.2.2 Use of Masking Agents. Use of masking agents (neutralizing agents and pleasant smelling agents) in landfill areas or near residential areas is a common approach to abate odor from the landfill. For example, terpenes can be added to discontinuous or small volumes of odorous emissions in order to overcome their environmental nuisance (Smet et al., 1998). There are a number of masking agents that are used for this purpose and in most cases privately owned. These masking agents are used to lower odor but not to remove odor compounds.

A few researchers have tested bacterial deodorants using sulfur oxidizing bacteria (Hirano, Kurosawa et al., 1996; Kurosawa et al., 1997). Snyder (2009) introduced a liquid deodorant composed of *Bacillus amyloliquefaciens* strain NRRL B-50141 with a range of concentration between 1×10^5/mL to 1×10^{10}/mL, propylene carbonate, sodium citrate, sodium bicarbonate, and sodium carbonate. Hirano et al. (1996) used *Thiobacillus thiooxidans* with sodium citrate buffer to remove H_2S and reported that the removal of H_2S (20 μL/L) in the gas phase ranged from 80% to 100% in the pH range of 2 to 7. A deodorant composition in concentrate form for abating the odor of organic refuse comprises isopropanol (solvent, 5–15% w/w) a glycol (antifreeze and a stabilizing agent, 2–8%), water (30–60%), an emulsifying and surfactant agent (20–40%), a binding agent (0.1–1.0%), and an essential oil (eucalyptus oil, lemon grass oil, and juniper berry oil, 5–15%; Roychowdhury, 1989). The inventors suggested that using 0.28–0.57 m^3 of a deodorant composition per acre of the landfill area would be effective for the abatement of landfill odor for a period of 3–5 days. Also, plant-derived oils were used to reduce odor from livestock manure (Varel, 2002). Carvacrol and thymol were found to inhibit odor generation by reducing anaerobic microbial activities. However, misting deodorant in landfills is a temporary solution for the odor reduction.

5. CONCLUSION

The control of H_2S from landfills is important for public health and environmental protection. The authors reviewed the current academic development on the formation of H_2S in landfills, the environmental risk of H_2S, the levels of H_2S in the landfill environment, and H_2S control. H_2S emission varies with waste composition, waste age, and environmental conditions for sulfur reducing bacteria, landfill geometry, and meteorological factors influencing gas emission.

Reducing sulfur sources in waste materials to be landfilled is the most effective option for lowering H_2S generation in landfills. When treating H_2S in collected landfill gas, physicochemical and biological technologies are commonly used. In addition to H_2S removal efficiency, cost is another most important consideration for choosing treatment technology. Physicochemical H_2S removal processes using absorption removal in iron-chelated solutions and iron-based adsorbents are mature technologies. Biological H_2S treatment technologies need high initial investments but their operating costs are lower than the physicochemical processes. The use of alternative cover is the most commonly used method to remove H_2S in fugitive landfill gas. Even though the range of H_2S adsorption capacity retention of the alternative cover materials varies with testing conditions, the alternative cover using recycled waste is a good option to the control surface emission of H_2S in landfills from both odor control and waste recycling perspectives.

It should be noted that although there are many methods available for H_2S control in landfills, the selection of technologies should be based on many considerations, such as waste composition, landfill location, climate condition, removal efficiency, and the initial and operational cost etc. In addition, other odor-causing compounds also need to be counted for odor control in landfills.

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Odor impact zones around landfills: Delineation based on atmospheric conditions and land use characteristics

Berrin Tansel *, Bahareh Inanloo

Civil and Environmental Engineering Department, Florida International University, 10555 West Flagler Street, Engineering Center, Miami, FL 33174, United States

ABSTRACT

Odors emitted from landfills can result in complaints by the residents living near the landfills. The aim of this study was to develop an assessment and delineation tool to identify the areas which can be impacted by the odors released from landfills based on land use characteristics and atmospheric conditions; and estimate the number of people who may be impacted. Odor emissions and dispersion analyses were conducted for three case study landfills under different atmospheric conditions in view of the land use characteristics around each landfill. Odor emissions and odor intensity levels were estimated based on the total gas production and the level of odorous compounds present in the landfill gas using the Landfill Gas Emissions Model (LandGEM) software. To delineate the odor impact zones, air dispersion characteristics of the odorous gases were analyzed using the dispersion modeling software, Areal Locations of Hazardous Atmospheres (ALOHA), and mapped using ArcGIS. Impact zone analyses were conducted based on the odor perception thresholds. The methodology developed involved coupling landfill gas emissions model (LandGEM), dispersions model (ALOHA) and mapping software for land use and population density (ArcGIS) allows visualization of the potential impact zones for preliminary delineation of the buffer zones around landfills, developing appropriate mitigation measures in view of the changing land use characteristics and population density around the existing and planned landfills. The odor impact zone delineation methodology was named Land-OZ (short for Landfill Odor Impact Zone). Results of using the odor impact zone tool showed that atmospheric stability could increase the odor impact radius around the three landfills evaluated between 340 and 1100 percent depending on the land use characteristics of the surrounding areas.

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1. Introduction

Odors are one the major causes of complains by the residents living around landfills. The odors emitted from landfills may cause temporary symptoms such as nausea and headache. Especially people with respiratory ailments (e.g., asthma) can be sensitive to the odorous compounds emitted from landfills. However, it is difficult to correlate these symptoms to specific compounds present in the air. It is difficult to establish effective measures for controlling emission and dispersion of odorous compounds from landfills due to lack of a direct correlation between the odor levels perceived by different individuals.

The type and intensity of odor emissions depend on the design and operation of the landfills; however, dispersion of odorous compounds depends on the land use, geographic characteristics, and atmospheric conditions around the landfills. Effective mitigation measures can be implemented by aligning the land development and landscape policies around the landfills to control dispersion of odorous compounds to neighboring areas and reduce odor related complaints by the communities around the landfills.

The aim of this study was to delineate the odor impact zones around active and closed landfills and quantify the impacted population in relation to atmospheric stability conditions and land use characteristics around landfills. Landfill Gas Emissions Model (LandGEM) was coupled with Areal Locations of Hazardous Atmospheres (ALOHA) dispersion modeling software. ALOHA (air dispersion model) was subsequently coupled with Geographic Information System (GIS) data to delineate the odor impact zone in terms of land use and population density under different atmospheric stability conditions. The odor impact zone delineation methodology developed was named Land-OZ, which is short for Landfill Odor Impact Zone.

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* Corresponding author.
   E-mail address: tanselb@fiu.edu (B. Tansel).

2. Odor emissions from landfills and odor dispersion

2.1. Odor regulations for solid waste facilities

In the United States, odor is not regulated by the U.S. Environmental Protection Agency as a pollutant. However, many states have established odor-based ambient air quality standards. The policies and regulations have different criteria which include health-based ambient air quality standards specifically for hydrogen sulfide (e.g., Massachusetts, North Dakota, and Ohio), controlling nuisance odors based on a dilution to threshold standard at or beyond the property boundary or odor level at the property boundary (e.g., Colorado, Connecticut, Delaware, Florida, Maine, Missouri, and Wyoming), and use of odor nuisance standards based on specific odor monitoring practices (e.g., Nevada, North Dakota).

In recent years, regulatory efforts have been directed towards characterization of volatile organic compounds (VOCs) originating from solid waste facilities due to their impacts on air quality and human health (i.e., psychological stress, irritation, toxic reactions) (Belpomme et al., 2007; Gallego et al., 2008; Hutter et al., 2006; Irigaray et al., 2007; Liang and Liao, 2007; Peng et al., 2006; Wolkoff and Nielsen, 2001). Regulatory agencies use general characterization for types and sources of the odor emissions for development of policies and regulations (TCEQ, 2007). Several methods have been used to assess and quantify odor emissions and odor annoyance levels. For landfills and composting facilities, chemical analyses are used to identify the key compounds during odor release episodes. Sensory methods (e.g., dynamic olfactometry and use of odor panels provide) are also used as a measure to establish the appropriate guidelines for odor annoyance levels (Belpomme et al., 2007; Gallego et al., 2008; TCEQ, 2007).

2.2. Characteristics of landfill gas

Landfill gas is generated during decomposition of organic material deposited (i.e., paper, animal, food waste, vegetable matter). During the biological decomposition process, the conditions in the landfill change from aerobic to anaerobic state as the availability of oxygen within the deposited waste layers declines. During the biological transformations, VOCs are formed in addition to methane (CH_4 45–60% v/v) and carbon dioxide (CO_2 40–60% v/v). Both methane and carbon dioxide are categorized as greenhouse gases (GHG) (Scheutz et al., 2009). The methane component in landfill gas has been reported to have a global warming potential (GWP) index of 25 over a 100-year time horizon (Lattanzio et al., 2016). On the other hand, the biogenic carbon dioxide emitted during waste decomposition is considered neutral for climate change (Manfredi et al., 2009; Capaccioni et al., 2011).

Municipal solid waste (MSW) landfills can be potential sources of air pollution and offensive odors, hence, affecting the quality of life and property values in the surrounding communities (Sarkar et al., 2003). The character (offensiveness) of an odor is a unique, inherent quality that does not vary with intensity. Odor intensity is the relative measure of the perceived concentration (TCEQ, 2007). Hence, offensiveness can be distinguished even at very low concentrations when the odorous compounds present in the air.

Both organic and inorganic sulfur compounds (i.e., hydrogen sulfide, sulfur dioxide, ammonia, and methyl mercaptan) have been identified as the primary causes for the odorous nature of landfill gas. The main compound for the strong odors originating from landfills is hydrogen sulfide (H_2S) (Pawlowska, 2014). Hydrogen sulfide is heavier than air, and may constitute up to 1% (v/v) of landfill gas emissions. Factors such as landfill size and age, environmental conditions in landfill (moisture, temperature, waste type, pH), and environmental conditions (climate, moisture, temperature, atmospheric conditions) influence the biodegradation processes and the odor emission potential of the landfills (Speight, 2011; Chemel et al., 2012). During waste decomposition, odorous gasses are produced even during the initial transition stage of the landfill as the oxygen gas is depleted. In general, after 20 years, majority of the organic wastes are decomposed; however, small quantities of gas may continue to be produced for 50 or more years after the landfill is closed (Chalvatzaki and Lazaridis, 2010).

2.3. Odor perception

Human nose can detect and distinguish odors at concentrations lower than the detectable thresholds by gas chromatography for some compounds (Rappert and Müller, 2005). Odor perception depends on physiology, weather conditions, seasonal changes, subjective perception, and interaction of different odors (Firestein, 2001; Noble et al., 2001; Davoli et al., 2003; Zou et al., 2003; Capelli et al., 2008; Gallego et al., 2008). How odors are processed at the behavioral and neural levels depends on past experiences, environmental conditions, and psychological and physical state of the individuals (Tansel et al., 2006).

In a recent landfill odor study, among the 68 odorous gases identified (inorganic compounds, halogenated compounds, aromatics, VFAs, aldehydes, ketones, esters, hydrocarbon, and other sulfur and nitrogen compounds); NH_3 and H_2S accounted for over 90% and 5% of the total odorous gas concentrations, respectively (Ying et al., 2012).

2.4. Challenges with odor monitoring programs at solid waste facilities

Transport of odorous compounds in air depends on the combined effects of climate conditions (e.g., temperature, humidity, air pressure, and wind direction), facility characteristics (e.g., landfill cover, landfill gas extraction method) and geographical location (e.g., elevation, tree cover, amount of precipitation). For waste management facilities, odor levels and odor emission rates cannot be easily determined by periodic sampling. Spot sampling/monitoring over a large area (e.g., landfill) make it questionable for collecting representative data due to variability of atmospheric stability and complexity of odor generation, air dispersion, and land use characteristics around the landfills which affect the odor perception of the people living in the vicinity.

2.5. Odor dispersion models and delineation tools

Different modeling approaches have been used to evaluate the odor impacts on receptors. Some dispersion models use peak-to-mean approach for odor dispersion. For example, the Austrian Odor Dispersion Model (AODM) uses a Gaussian dispersion approach with peak-to-mean factors depending on wind speed and atmospheric stability (Piringer et al., 2012). On the other hand, The German Lagrange code AUSTAL2000 uses a constant factor 4 in all meteorological conditions to derive the maximum odor concentration over a short integration time. The Lagrange model can be applied in complex topography and with isolated buildings while the Austrian peak-to-mean approach in AUSTAL2000 allows for more realistic separation distances in these environments (Piringer et al., 2012). Capelli and Sironi (2018) compared two odor dispersion approaches; one that considers emissions as a function of the wind speed and the other one that considers odor emissions at constant rate. The comparison of the field assessments showed that the first approach significantly overestimated the landfill odor emissions, whereas the use of a constant odor emission rate had better agreement with the field data. Li and Guo (2006) developed a three-dimensional model which incorporates atmospheric stabil-

ity and wind, and temperature vertical profiles in the atmosphere. Using a Lagrangian discrete phase model driven by a large-eddy simulation (LES) turbulent flow field, they predicted downwind odor concentration. They also compared the CFD modeling results with those obtained by the CALPUFF model, a Lagrangian puff model recommended by the U.S. Environmental Protection Agency. Both models showed that odor traveled farther under stable conditions than under unstable conditions with the same wind speed. Under the same atmospheric stability category, odor concentrations were higher at lower wind speed than that at higher wind speed. Stable atmospheric conditions and lower wind speeds resulted into higher odor concentrations and longer travel distances.

3. Methodology

To identify the odor impact zones around landfills, three software (for modeling and mapping) were used interactively to estimate the rate of odor emission, atmospheric dispersion of odorous compounds in relation to atmospheric stability, and mapping of the impact zones to delineate the impacted areas and estimate the affected population within the impact zones.

3.1. Emission estimates

The odor levels were estimated based on the total gas production and the concentration of the odorous compounds present in the landfill gas using the LandGEM. To delineate the odor impact zones, the emission rates of the odorous compounds were analyzed using air dispersion model ALOHA and mapped using ArcGIS. ALOHA uses the "threat zone" terminology for chemical releases where the concentrations in air exceed a critical value. For the odorous compounds, this area was referred as "odor impact zone" in this study. Fig. 1 presents the overall framework used for the analyses to delineate the odor impact zones. The final odor impact zone delineation methodology developed was named Land-OZ, which is short for Landfill Odor Impact Zone.

LandGEM uses Microsoft Excel interface to estimate the emission rates from landfills. It was developed by the Clean Air Technology Center of the US Environmental Protection Agency (US EPA, 2005). The software estimates the methane generation using the following first order exponential equation:

$$Q_{CH4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} kL_0(M_i/10)(e^{-kt_{ij}}) \tag{1}$$

where Q_{CH4} is the estimated methane generation flow rate (cfm); i and j are 1-year and 0.1-year time increments, respectively; n is the difference between year of calculation and initial year of waste acceptance; k is the methane generation rate (1/year); and t_{ij} is the

age of the jth section of waste mass buried in the ith year (decimal years); L_0 is the potential methane generation capacity (ft^3/ton), M_i is the mass of solid waste disposed in the ith year (ton) in the landfill. LandGEM allows the input of site-specific values for methane generation rate (k) and potential methane generation capacity (L_0). In the analyses of the case study landfills, default values provided by LandGEM for the conventional landfills in the US were used. These values were 0.04 per year for the decay rate constant (k) and 100 m^3/Mg for the potential methane generation capacity (L_0).

Based on the total gas generation and the composition of landfill gas; the generation rates of other compounds can be estimated in relation to their concentration in the landfill gas. Table 1 presents the typical composition of landfill gas. In this study, hydrogen sulfide was used as the key compound (i.e., main odor source from landfill) for odor emission analyses. The emission rates of hydrogen sulfide were estimated based on the concentration of hydrogen sulfide present in the landfill gas and the quantities of landfill gas emissions estimated by LandGEM. Hydrogen sulfide is heavier than air; therefore, it would be transported at the lower levels of the atmosphere. Using the perception concentration threshold of hydrogen sulfide by humans, odor impact zone around the landfills were delineated by assuming that the landfill is a continuous source for hydrogen sulfide emissions. Table 2 presents the odor threshold levels for the volatile compounds in the gas generated at the municipal solid waste landfills (ATSDR, 2015). For the model, average odor thresholds were used for the odor threshold ranges.

It should be noted that, LandGEM uses a simple approach (first-order decomposition kinetics) for quantifying emissions during decomposition of waste from landfills. Changes in operating practices over time can impact the emissions potential of MSW. In the engineered landfills, only a small fraction of the volatile organic compounds produced would emitted to the atmospheric environment. In case of closed landfills, the cap and the cover soil would reduce the emission rate. In the case of active landfills, only working area are exposed to the air, hence, the use of total gas production may overestimate the odor impact. Therefore, the values provided by LandGEM should be considered as the preliminary estimates and should be adjusted in view of the conditions of the landfill for delineating the odor impact zones.

3.2. Atmospheric dispersion

The odor impact zones were estimated conservatively by assuming that hydrogen sulfide present in the landfill gas would be emitted from the entire landfill and the buffer zones were delineated from the boundary of the landfill site. Since the atmospheric conditions (i.e., temperature, humidity, solar radiation, and cloud
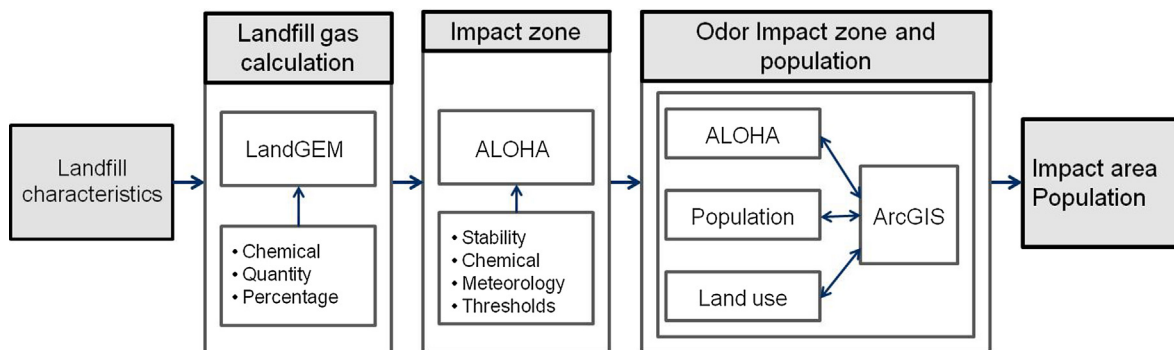

Fig. 1. Framework for the odor impact zone delineation from landfills.

Table 1
Composition of landfill gas (ATSDR, 2015).

Component	Composition (% v/v)
CH_4	45–60
CO_2	40–60
N_2	2–5
O_2	0.1–1.0
NH_3 [c]	0.1–1.0
NMOCs [a]	0.01–0.6
Sulfides [b,c]	0.0–1.0
Hydrogen	0.0–0.2
CO	0.0–0.2

[a] Non-methane organic compounds.
[b] Hydrogen sulfide, dimethyl sulfide, mercaptans.
[c] Odorous.

Table 2
Landfill gas components odor thresholds (ATSDR, 2015).

Component	Odor threshold (ppb)
Hydrogen sulfide	0.5–1.0
Ammonia	1000–5000
Benzene	840
Dichloroethylene	85
Dichloromethane	205,000–307,000
Ethylbenzene	90–600
Toluene	10,000–15,000
Trichloroethylene	21,400
Tetrachloroethylene	50,000
Vinyl chloride	10,000–20,000

cover) affect the dispersion of the odorous compounds, three different atmospheric stability conditions were analyzed as representative conditions.

The air dispersion model used, Areal Locations of Hazardous Atmospheres (ALOHA), was developed for evaluating dispersion of hazardous chemical vapors, including toxic gas clouds, fires, and explosions (NOAA, 2005). Based on the release characteristics of the compounds, the model estimates the impact zones (referred as threat zones) where the concentration of the chemicals in air are expected to exceed the levels specified by the user (US DOE, 2004).

The air dispersion model (ALOHA) was used to visualize the odor impact zones around the landfills based on the size and operational characteristics (e.g., accepted annual waste) of the landfills and atmospheric conditions. The residents living around the landfills do not experience the same level of odor intensity. The odors often reach the human detection threshold during certain times of day (e.g., early mornings) or under specific weather conditions (e.g., high humidity and after rain) which are directly influenced by the atmospheric stability conditions.

ALOHA provides estimates of how quickly a chemical will escape and form a gas cloud, and how the gas cloud will travel downwind. A series of dialog boxes prompt user to enter information about the emission scenario (e.g., chemical, weather conditions, and the type of release). The software maps the estimated threat zone corresponding to the area where a particular hazard (e.g., toxicity, concentration level) is predicted to exceed a user-specified level of concern (LOC) after the release. However, ALOHA's concentration estimates can be less accurate when the conditions are such that mixing is reduced in the atmosphere, for example at very low wind speeds (<3 miles per hour) and under very stable atmospheric conditions (stability classes E and F) which generally occur at night or in the early morning (e.g., low-lying fog). Under these atmospheric conditions, gas concentrations within the pollutant cloud can remain high far from the source.

3.3. Impact zone and impacted population

The inputs for the dispersion analyses include wind direction, wind speed, humidity, date and time, temperature, location and chemical characteristics for the odorous compounds. For the analyses, it was assumed that the total landfill gas produced is emitted into the atmosphere and the dispersion calculations were performed using the maximum emission rate. The estimated odor annoyance distance was delineated as the buffer zone around the landfill using ArcGIS software. In the calculations, a wind speed of 5 miles per hour was used to compare the magnitude of the impact radius around the case study landfills at the baseline wind speed condition. A constant value of 5 miles/hour was used for wind speed. This wind speed corresponds to fresh breeze conditions in Beaufort wind scale (i.e., sway of small trees and breaking of waves on inland waters). The wind speed was used in the analyses because it represents relatively conservative estimates for delineation of the odor dispersion zone and at very low wind speeds the dispersion estimates by ALOHA is not accurate due to cloud/fog formation which traps the emitted gases and interferes with the dispersion estimates. This wind speed is typical of the most prevalent conditions during normal operational conditions at landfills. The atmospheric stability conditions corresponding to this wind speed were selected based on the guidelines provided by Turner (1970) and Hunter (2012) and as presented in Table 3. The stability conditions B (unstable) and C (neural) were evaluated for the day time and stability condition F (stable) was evaluated for the night time dispersion characteristics.

In this study wind direction was not taken into account in the analyses. Use of site specific prevalent wind conditions would identify directions downwind from the landfill to identify areas that would be affected by the emissions originating from the landfill most of the time versus upwind areas which may be affected occasionally. The analyses aimed to identify areas which can be potentially impacted around the landfills (wind blowing from any direction), as the wind direction changes during the day. The population within the impact zones was estimated using ArcGIS and population density maps.

4. Case study landfills

Three case study landfills were used to demonstrate the methodology. These landfills were Okeechobee Landfill in Florida, South Dade Landfill in Florida, and Fenimore Landfill in New Jersey. Table 4 presents the general characteristics of the case study landfills. Okeechobee landfill is the largest of the three landfills in terms of both the area and municipal solids waste (MSW) deposited per year. It is located in a rural area in Central Florida, USA, with low population density. The Fenimore landfill is the smallest of the three landfills, but it is located close to the residential areas. This landfill has drawn attention because of the numerous complaints filed by the neighboring communities for the odor emissions. Okeechobee and South Dade landfills are both operational and active landfills. The Fenimore landfill was closed in 1979 but reopened temporarily in 2012.

5. Results and discussion

Odor impact zones were delineated around each landfill for different atmospheric stability conditions corresponding to unstable, neutral and stable conditions; using the atmospheric stability conditions B, C and F, respectively, as defined by the Pasquill-Gifford stability categories (Hunter, 2012). Stabilities B, and C refer to daytime hours with unstable conditions. Stability F refers to nighttime and stable conditions and it is based on the amount of cloud cover.

Table 3
Pasquill-Gifford stability classes (Turner, 1970; Hunter, 2012).

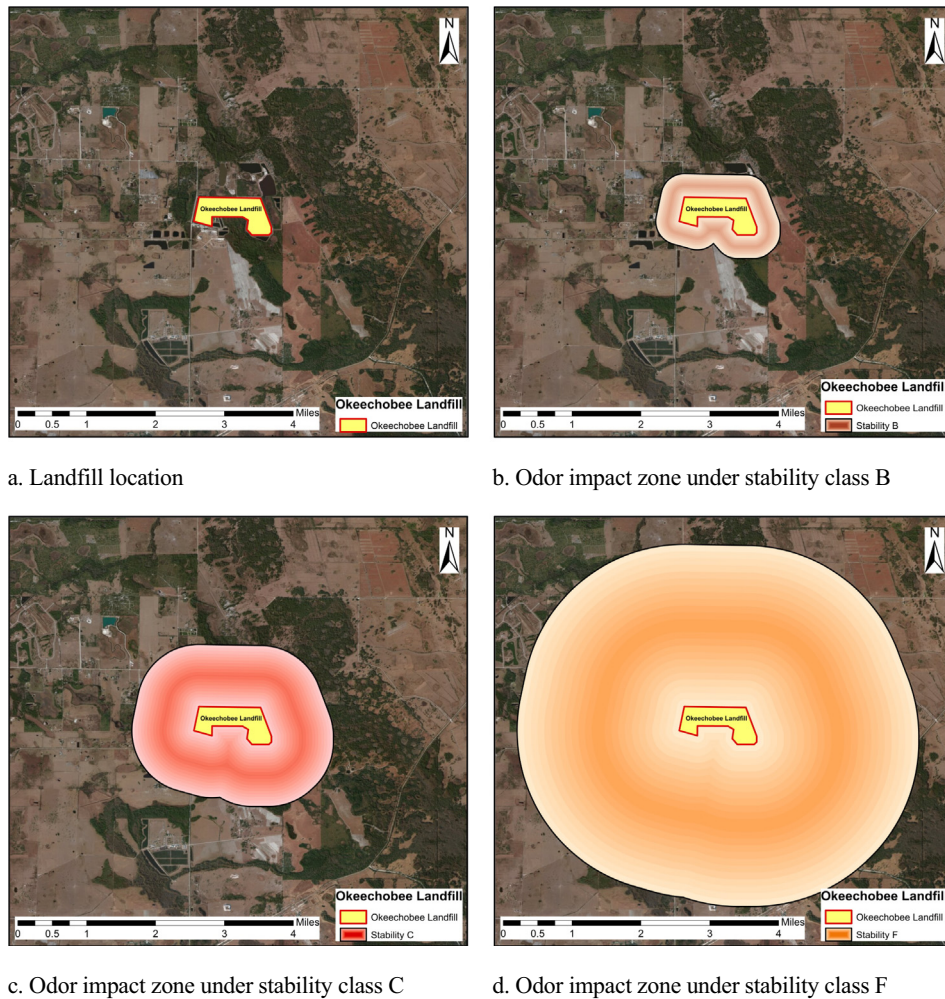| Surface wind speed		Day time incoming solar radiation			Night time cloud cover	
(m/s)	(mi/h)	Strong	Moderate	Slight	>50%	<50%
<2	<5	A	A–B	B	E	F
2–3[a]	5–7 [a]	A–B	B [a]	C [a]	E	F [a]
3–5	7–11	B	B–C	C	D	E
5–6	11–13	C	C–D	D	D	D
>6	>13	C	D	D	D	D

[a] Conditions used for the numerical analyses.

Table 4
Case study landfills.

Landfill name	Location	Year of opening	Year of closure	Area (Acres)	MSW deposited (Tons/year)
Okeechobee	Florida, USA	1992	2052	833	1,600,000
South Dade	Florida, USA	1979	2029	300	600,000
Fenimore	New Jersey, USA	1950	1979	109	400,000

Figs. 2–4 compare the odor impact zones around the case study landfills under different atmospheric stability conditions. The color gradation in the figures is due to the curvature of the odor dispersion profile (i.e., spherical or near spherical shape centered at the source). This allows the model to be implemented for 3D visualization.

The odor impact zones showed different impact radii due to differences in characteristics of the landfills (e.g., size of landfill and amount of solid waste deposited) and land use in the neighboring areas. For the unstable atmospheric conditions (typical for afternoon; conditions B and C), the mixing layer is higher; hence, the

a. Landfill location

b. Odor impact zone under stability class B

c. Odor impact zone under stability class C

d. Odor impact zone under stability class F

Fig. 2. Odor impact zones for Okeechobee landfill.

a. Landfill location

b. Odor impact zone under stability class B

c. Odor impact zone under stability class C
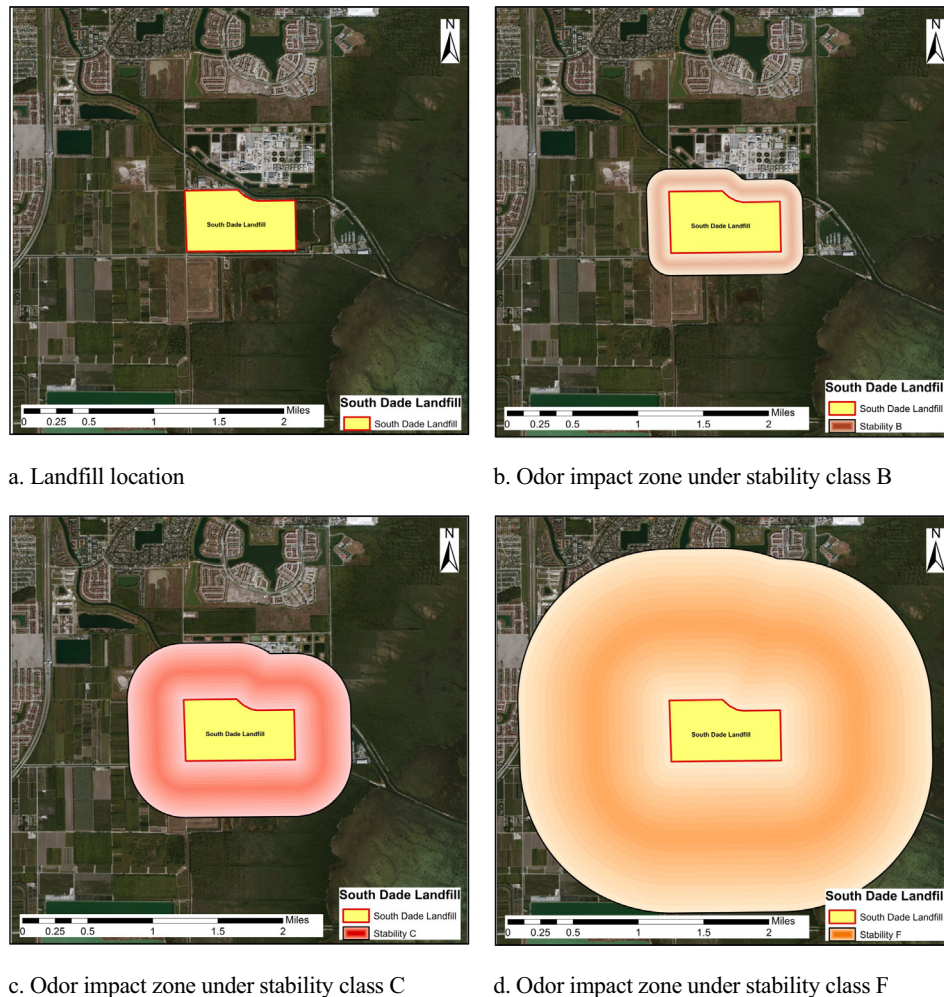
d. Odor impact zone under stability class F

Fig. 3. Odor impact zones for South Dade landfill.

odorous gases released would be mixed by vertical air movements, keeping the impact radius smaller. However, under stable atmospheric conditions (typical for nighttime, condition F), the mixing depth is lower; hence, air moves closer to the ground and the odorous compounds can be carried over larger distances. However, for the stability condition F, the odor strength would be reduced since the air would be diluted over a larger volume. Table 5 compares the odor impact radii for the three case study landfills under different atmospheric stability conditions. The diurnal changes in temperature in the lower atmosphere due to heating and cooling of the ground surface affects the stability of the atmosphere. In general, cooling from below (typically occurring during night time) promotes stability, while heating from below (typically occurring during day time) creates instability. The stability conditions are representative of conditions during day time (B unstable and C neutral) and during night time (F stable). It should be noted that dispersion characteristics of odorous compounds in air depend on both the atmospheric conditions and the distance downwind from the landfill sites. During day time hours the odor impact radius would be smaller; however, during the night time hours the odor impact radius could be between five to ten times larger depending on the terrain (Table 6).

The increased surface roughness in urban areas (e.g., high rise buildings, tree coverage) enhances the turbulence and dispersion as well as the heat flux. On the other hand, roughness decreases wind speed. In urban areas, the air with the odorous compounds

would move slowly; however, the air would be mixed over a smaller area resulting in longer exposure time for the population in these areas. Typical surface roughness values are 0.002–0.02 m for short grass cover, 0.04–0.2 m for pasture land, 0.05–0.06 m for suburban housing, and 1–5 m for forests and cities. Stable air from the rural areas becomes unstable as it moves over the warmer urban areas (US EPA, 1999). In general, as the air moves downwind, the odorous compounds in the air are dispersed over a larger area, hence, the concentration decreases. However, in urban areas, the air (and contaminants) would be trapped between the structures and trees due to higher surface roughness which causes stagnant conditions. For the surface roughness, mid values of the surface roughness ranges were used corresponding to the current surface characteristics of the surrounding areas of the landfills.

The presence of large open areas and lack of good tree coverage around the Okeechobee landfill allow air to be dispersed over larger distances. However, the areas around this landfill is sparsely populated, hence, the affected population is relatively small (Table 7). On the other hand, South Dade landfill is closer to the residential areas with good tree coverage. Therefore, the odor impact radius is almost half of that for the Okeechobee landfill. However, the affected population near the South Dade landfill is more than twice that of the Okeechobee landfill (Table 7).

The Fennimore landfill has both residential communities and open areas around the site. Development of the areas around the landfill since its closure in 1979 increased the population density
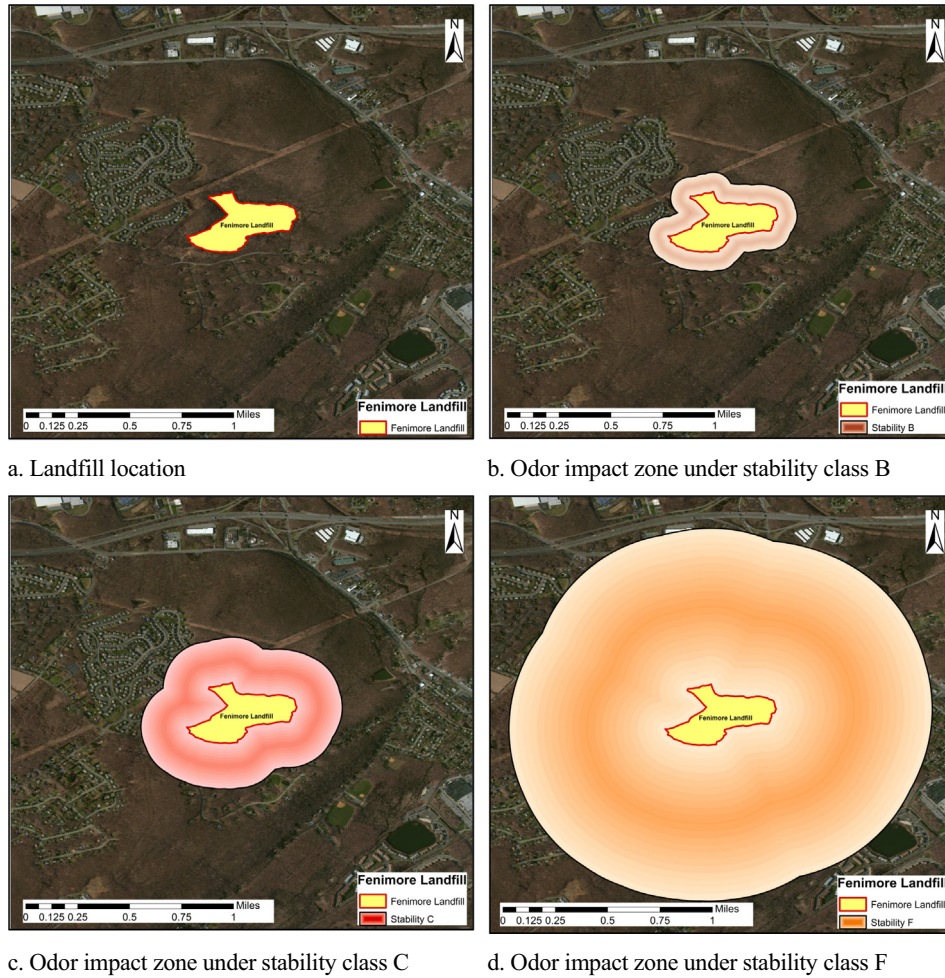
a. Landfill location

b. Odor impact zone under stability class B

c. Odor impact zone under stability class C

d. Odor impact zone under stability class F

Fig. 4. Odor impact zones for Fenimore landfill.

Table 5
Comparison of estimated impacted areas and population by odor emissions around the case study landfills for atmospheric stability conditions corresponding to unstable (B), neutral (C) and stable atmospheric conditions (F).

Landfills	Impact radius (miles)			Highest odor strength zone (miles)		
	B	C	F	B	C	F
Okeechobee, FL	0.5	0.8	2.2	0.20–0.30	0.30–0.50	0.90–1.30
South Dade, FL	0.1	0.4	1.2	0.04–0.06	0.10–0.30	0.40–0.80
Fenimore, NJ	0.08	0.2	0.8	0.03–0.05	0.08–0.12	0.30–0.50

Table 6
Increase in the impacted area and population when atmospheric stability changes from B (unstable) to C (neutral) during day time and from B (unstable) during day time to F (stable) during night time.

Landfills	Odor impact radius (% increase)		Impacted area (% increase)		Impacted population (% increase)	
	B to C	B to F	B to C	B to F	B to C	B to F
Okeechobee	60	340	194	1161	300	1279
South Dade	300	1100	147	686	247	786
Fenimore	150	900	123	985	223	1085

around the landfill. Also, the large open areas around the site serve as corridors for air movement, allowing the odorous compounds to travel larger distances with air currents. Although the Fenimore landfill has the smallest impact area, it has the highest affected population due to the residential land use (i.e., high population density) around the site.

At landfill sites which may have snow cover or freezing soils during winter periods, atmospheric stability classifications may need to be reevaluated. However, the odor release potential during the winter months would be relatively reduced due to lower biodecomposition rates at lower temperatures. It would be more suitable to conduct the odor impact evaluations for the conditions

Table 7
Comparison of estimated impacted areas and population by odor emissions around the case study landfills for atmospheric stability conditions corresponding to unstable (B), neutral (C) and stable atmospheric conditions (F).

Landfills	Impact area (Square miles)			Impacted population		
	B	C	F	B	C	F
Okeechobee	1.97	5.80	24.85	28	84	358
South Dade	0.96	2.37	7.55	107	264	841
Fenimore	0.26	0.58	2.82	270	603	2930

that are prevalent during the warmer and rainy seasons as the odor release potential would be higher during these conditions. The release of biogas from landfills can be reduced by waste separation, daily cover, incorporating odor neutralizing (or odor absorbing) materials into daily cover, and limiting exposure to rain and wind exposure during waste deposition activities by keeping the active face as small as possible.

6. Conclusions

Delineation of odor annoyance zones can be an effective planning tool for developing appropriate mitigation measures around landfills to minimize odor complaints. Although many states have developed odor-based ambient air quality standards and monitoring programs, these requirements may not provide an effective measure based on changes in land use and population density and dynamics of the atmospheric conditions around landfills. A simple methodology was developed to delineate the odor impact zones around landfills by interactively coupling LandGEM, ALOHA, and ArcGIS softwares. Analyses conducted using the methodology developed (Land-OZ) showed that land use characteristics around landfills significantly affect the dispersion characteristics and odor impact radius, hence, number of people affected by the odor emissions. Atmospheric stability conditions (e.g., day time and night time conditions) also can have a significant effect on odor dispersion characteristics. For example, stable atmospheric conditions at nighttime (condition F) would result in larger impact radius; however, odor strength would be less since the air would be diluted over a larger volume.

It should be noted that the dispersion model used (ALOHA) has some limitations as it was developed for emergency response to spills. Corrections and modification should be considered based on knowledge and experience with the regional conditions (e.g., prevalent wind direction), operating conditions at the landfills, emission potential of odorous gases (from working face and/or through cap), and land use characteristics around the landfills. In view of the prevalent wind direction (during day and night as well as different seasons), allocating adequate buffer zones as well as landscape policies in down wind direction should be considered to mitigate potential odor impacts around landfills.

The methodology developed in this study (by coupling readily available software) can be used as a planning tool for preliminary delineation of buffer zones around landfills, developing appropriate mitigation measures for landfill operation and landscape policies, evaluation of regional land use plans, and current and projected population density around existing and planned landfills. Although landfill odor is not classified as a health hazard, the discomfort caused by odorous emissions can interrupt daily activities of the residents around the landfill sites.

Acknowledgements

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