Journal of Environmental Science and Technology

ISSN 1994-7887

ANSInet
an open access publisher
http://ansinet.com

Journal of Environmental Science and Technology 9 (1): 1-25, 2016
ISSN 1994-7887 / DOI: 10.3923/jest.2016.1.25

A Review on Biogas Interception Processes in Municipal Landfill

[1]Ahmad Qasaimeh, [2,3]Mohammad Raji Abdallah/Qasaimeh and [2]Falah Bani Hani

[1]Department of Civil Engineering, Jadara University, Jordan

[2]Department of Chemical Engineering, Al-Huson University College, Al-Balqa' Applied University, Salt, Jordan

[3]Departments of Civil and Environmental Engineering, College of Engineering at Al Lith, Umm Al-Qura University, Saudi Arabia

Corresponding Author: Ahmad Qasaimeh, Department of Civil Engineering, Jadara University, Jordan

ABSTRACT

Biogas in landfill is being captured by natural and engineered processes. The natural processes are represented by biological activities such as bacterial methane oxidation and plant uptake for carbon dioxide at topsoil layer. Landfill gas is transported through soil layers in landfill top or in nearby areas before being released to the atmosphere. Whilst transported in the soil layers the biogas is mixed with atmospheric air and the methane may hence be oxidized by the methanotrophic bacteria in the soil using oxygen from atmosphere. Methane oxidation is affected by different environmental factors such as; temperature, water content, nutrients, substrate and oxygen concentrations. One of the ways to decrease greenhouse emissions in the future is to plant fast growing woody crops thereby sequestering carbon and displacing fossil fuels by harvesting woody biomass for bio-energy, or by storing carbon in long-lived woody products. Plant uptake for carbon dioxide is affected by some parameters such as; CO_2 concentration, nitrogen concentration, water content and temperature. The engineered processes are represented by various physical biogas extractions; gas is collected using network of collection pipes and wells. The gas collection efficiency in landfills is between 40-90%. Landfill gas can be collected by either a passive or an active collection system. Passive gas collection systems use existing variations in landfill pressure and gas concentrations to vent landfill gas into the atmosphere or a control system. Active gas collection is considered a good means of landfill gas collection. An active collection system composed of extraction wells connected to header pipe to a pump that delivers gas for energy recovery.

Key words: GHGs, biogas, methane oxidation, CO_2 plant uptake, gas extraction

INTRODUCTION

Solid waste at landfills is a free source of energy, biogas is generated naturally by bacterial activity and thus municipal solid waste is recently manipulated via several approaches that entail sustainable management systems, techniques, models, policies and strategies (Liamsanguan and Gheewala, 2008; Rada *et al*., 2009; Ionescu *et al*., 2013; Vaccari *et al*., 2012).

The biodegradable municipal solid waste material is made up of vegetation, domestic matter, paper, garden waste, wood and textile material. The organic carbon is the common element between biodegradable materials that allows the development of energy and methane production. On previously mentioned bases, municipal waste is being investigated by many studies (Trulli *et al*., 2013; Chanakya *et al*., 2009; Rada *et al*., 2013).

In land filled waste, the biochemical reactions go through several phases: hydrolysis phase, an acidogenesis phase and the methanogenesis, which is the main source of methane contained in the final biogas mixture. Biological processes under anaerobic conditions drive the production of biogas, a gaseous mixture composed almost of methane (CH_4) and carbon dioxide (CO_2). Anaerobic bacteria is responsible of the decomposition of organic material (Davis and Cornwell, 2008).

Landfill gas production rate is influenced by functional parameters such as abiotic factors and landfill operation procedures (Christensen *et al*., 1996). The abiotic factors of concern can be summarized as: pH, nutrients, inhibitors, temperature and water content, while the landfill operation procedures of concern can be summarized as; waste composition, sewage sludge addition, shredding, compaction, soil cover, recirculation of leachate and pre-composting.

The methane producing bacteria is called methanogens which operate within 6-8 pH in methanogenesis process. Anaerobic ecosystem requires much less nitrogen (N_2) and phosphorous (P) than the aerobic system does. The optimal ratio for organic matter (as COD), N_2 and P content, is 100:0.44:0.08 (Christensen *et al*., 1996). Nutrients and metal supplementation have a positive effect on biogas production rate (Isci and Demirer, 2007). An adequate ratio of nitrogen (N) and phosphorus (P) is also required by the anaerobic ecosystem (Zhang *et al*., 2014).

The temperature for mesophilic bacteria is in the range of 30-35°C, while 45-65°C is for the thermophilic bacteria. The active temperature for methanogens is in the range of 30-50°C. The optimum temperature range of gas generation is between 30-45°C during the main landfill gas generation phase (Williams, 2005). The change of temperature will have an impact on the growth and the activity of the microorganisms (Mora-Naranjo *et al*., 2004). The range of moisture content in a typical landfill is 15-40% with a typical average of 30% (Williams, 2005). The moisture content controls methane production, since it stimulates microbial activity by providing closer contact between soluble and insoluble substrates and bacteria (Machadoa *et al*., 2009).

There are two main ways to capture biogas emission from landfills. One, like those in Northern Europe where there are many small and old landfills with low gas generation that is biologically uptaken (Jones and Nedwell, 1993; Boeckx *et al*., 1996; Borjesson and Svensson, 1997), where biological biogas uptake at landfills is apparently executed by biological methane oxidation by methanotrophs and carbon dioxide uptake by plants. Other option is gas collection and utilization, which could be very effective with a high gas generation. In this case, biogas can be collected by means of vertical and horizontal drain pipes and is employed to produce heat and energy. There are different technologies being studied to find the best handling of biogas collection (Andreottola and Cossu, 1988; EMCON Associates, 1980; El-Fadel, 1991).

BIOLOGICAL METHANE OXIDATION

Methane has Global Warming Potential (GWP) 21 times more than carbon dioxide. Atmospheric methane concentration has doubled during the last 100 years and is an explosive gas in concentrations between 5 and 15% in air, a 10% of global anthropogenic source of methane is from MSW landfills (Chalvatzaki and Lazaridis, 2009; Zhang *et al*., 2008).

While soils have not been considered as significant sinks for methane until recently, methane consumption has been reported in agricultural soils, forest soils, tundra and bogs (Topp and Hanson, 1991). Biological oxidation of CH_4 by bacterial methanotrophs has attracted much attention from the research community as an inexpensive waste gas treatment mechanism. Methane oxidizing activity, with a decrease in soil oxygen and an increase in microbial biomass, has been demonstrated in soils around leaks in natural gas pipes (Adams and Ellis, 1960) and in landfill covers (Kightley *et al*., 1995; Whalen *et al*., 1990; Bogner *et al*., 1995). Methane oxidizing activity in soils is an event that could have a strong effect on CH_4 emissions control from sources such as municipal landfills and the optimization of this process may give out an inexpensive strategy for controlling and utilizing emissions of this potent greenhouse gas.

Microbial oxidation in well-drained soils is the only identified biological sink for atmospheric CH_4 and accounts for 3-9% of total annual atmospheric CH_4 destruction (Prather *et al*., 1996). This is similar in magnitude to the current atmospheric increase (Houghton *et al*., 1996). Accordingly, alterations of the soil sink strength are a significant determinant of the rate of change in the atmospheric CH_4 concentration (Prather *et al*., 1996) and absence of this sink will cause the atmospheric CH_4 concentration to increase at 1.5 times the current rate (Duxbury, 1994). The control of aerobic methane oxidation is obviously related to the requirement for oxygen and methane. As a result, maximum oxidation rates are found where diffusion of oxygen from above and of methane from below is optimal for methanotrophs (King, 1992; Sundh *et al*., 1995a, b).

Xin *et al*. (2004) showed that methanotrophs oxidizes methane to carbon dioxide through sequential reactions catalyzed by a series of enzymes including methane monooxygenase, methanol dehydrogenase, formaldehyde dehydrogenase and formate dehydrogenase. Methanotrophic bacteria cultivate aerobically on methane as a sole source of carbon and energy. The first two enzymes involved in methane oxidation are methane monooxygenase (MMO) and methanol dehydrogenase (MDH) (Anthony, 1986). The MMO oxidizes methane to methanol and MDH catalyzes the oxidation of methanol to formaldehyde. *Methylosinus trichosporium* OB3b is a methanotrophic bacterium and contains two forms of MMOs: a soluble (sMMO) and a membrane-bound particulate (pMMO) whose syntheses depend on growth conditions (Murrell *et al*., 2000; Nielsen *et al*., 1997; Takeguchi and Okura, 2000).

Methane emission rates were strongly dependent on the oxidation coefficient utilized, which varies with the cover material and microclimate conditions of the site (Moreira *et al*., 2015). The maximum methane oxidation rates values for compost cover layer is higher than regular soil cover (Abichou *et al*., 2009). Some experiments in the literature such as, Im *et al*. (2009), Abichou *et al*. (2009), Scheutz *et al*. (2008, 2009) and Scheutz and Kjeldsen (2004) show that methane oxidation rate in landfill covers obtained in field and laboratory measurements usually formatted as parameters follow Michaelis-Menten kinetics equation and this present very different rates of methane oxidation under different site and microclimatic conditions. Methane oxidation rates depend upon different variables such as microclimate conditions, temperature, atmospheric pressure, moisture, soil conditions, oxygen concentration, methane concentration and amount of methanotrophs (Spokas and Bogner, 2011; Chanton *et al*., 2011).

Landfill gas is transported through soil layers in landfill top covers or in nearby areas before being released to the atmosphere. Whilst transported in the soil layers the biogas is mixed with atmospheric air and the methane may hence be oxidized by the methanotrophic bacteria in the soil using oxygen from atmosphere. Methane oxidation is affected by different environmental factors such as: temperature, water content, nutrients, substrate and oxygen concentrations (Hanson and Hanson, 1996). In the following sections is the description of environmental factors effect on methane oxidation.

Effect of soil moisture: Christophersen *et al*. (2000) showed that methane oxidation rate is a function of soil moisture content at different temperatures for the different soils. The optimum soil moisture content was different for each soil and depended on the temperature. Some soils had the lowest optimum soil moisture content and others had the highest. At lower temperatures the difference in oxidation rates with soil moisture content was reduced.

Methane oxidation rates decreased extensively after soil samples were dried below field moisture contents, increased to an optimum value as water was added and decreased with sustaining water addition. The maximum oxidation rate occurred at moisture content of 15.4% (dry weight basis) (Stein and Hettiaratchi, 2001). The texture and structure of soil will influence its moisture content in a manner that is site specific, depending on climatic variables such as temperature, solar flux, average wind speed and the type of vegetative cover (Stein and Hettiaratchi, 2001). Methane uptake was controlled strongly by soil moisture, with reduced fluxes under conditions of very low or very high soil moisture contents. The mineral soil Q10 (Q10 is the value for how many times the oxidation rate increases when temperature is increased 10°C at temperatures below the optimum temperature) of 1.11 for CH_4 uptake indicates that methane uptake is controlled primarily by physical processes (Bowden *et al.*, 1998).

Effect of temperature: Christophersen *et al.* (2000) showed for all the soil investigations, the oxidation rate increased with increasing temperature. Predictably, optimum temperatures were not found in this experimentation. For all the soils the optimum temperature must be higher than 15°C, which was the highest temperature in these explorations. Most researchers found optimum temperatures around 30°C, which do seldom occur in temperate soils. Dunfield *et al.* (1993) found optimum temperature for the methane oxidation around 25°C. As the temperature is increased, CH_4 oxidation increases exponentially to maximum and then decreases with continued temperature increase (Stein and Hettiaratchi, 2001).

At high methane concentrations the oxidation becomes saturated and the rate-limiting stage is the enzymatic action. Thus, the temperature response is something like parabolic: increasing rates with increasing temperature to a maximum and declining with continued temperature increase (Bailey and Ollis, 1986). King and Adamsen (1992) investigated soil cores where the temperature was increased between 0 and 30°C. The depth distribution of methane consumption and methane diffusion showed low sensitivity to changes in temperature. They observed methane consumption at -1°C and they suggested that methane consumption might occur at low temperatures on condition that the soil water remains liquid. Sommerfeld *et al.* (1993) showed that the soil microflora was active even when the soil was snow-covered and near 0°C and that methane consumption was taking place under that condition. In the 0-10°C range methane oxidation was about 13-38% of maximum activity. Prieme and Christensen (1997) observed methane oxidation to be active at low temperatures, down to 1°C in the field and -2°C in soil core experiments. Both in the field and in soil cores similar temperature responses of methane oxidation were measured. This indicated that temperature acts directly (i.e., via its effect on enzymatic processes and methane diffusion) on methane oxidation in the field. They suggested that the small temperature response of methane oxidation was partly a result of low substrate concentration.

Effect of organic matter content and methane concentration: Oxidation rates increased with increasing organic matter content. The optimum soil moisture content also increased with increasing organic matter content (Christophersen *et al.*, 2000). Oxidation of methane in top covers of landfills has been observed on several occasions and soil exposed to elevated methane concentrations can develop a high potential for methane oxidation (Whalen *et al.*, 1990;

Kightley *et al*., 1995; Boeckx *et al*., 1996; Czepiel *et al*., 1996; Borjesson and Svensson, 1997; Bogner *et al*., 1997). Low initial methane concentrations resulted in low maximal oxidation rates (Boeckx and van Cleemput, 1996; Boeckx *et al*., 1996). The oxidation rates at low initial methane concentrations were much lower than oxidation rates at high initial methane concentrations and the oxidation was performed by different kinds of bacteria (Bender and Conrad, 1992). Several researchers have shown that different species of bacteria are active at low and high methane concentrations (Bender and Conrad, 1994; Kightley *et al*., 1995; Bogner *et al*., 1997).

Effect of nitrogen content: There is a worldwide increase in atmospheric nitrogen (N) deposition on terrestrial and aquatic environments (Matthews, 1994; Galloway *et al*., 1995). Methanotrophs are inhibited by high soil N; driving attention that the contemporary worldwide increase in atmospheric N deposition will decrease soil CH_4 oxidation. Oxidation of CH_4 by methanotrophic and methylotrophic bacteria occurs in aerobic soils and the magnitude and rate of oxidation are influenced by soil type, aeration, environmental parameters and Nitrogen availability (Topp and Pattey, 1997; Le Mer and Roger, 2001). Application of fertilizer has been shown to inhibit CH_4 oxidation in soil (Steudler *et al*., 1989; Hutsch, 1998; Tlustos *et al*., 1998; Kravchenko *et al*., 2002). Nitrogen content frequently shows low atmospheric CH_4 oxidation comparative to unfertilized controls (Steudler *et al*., 1989; King and Schnell, 1994; Sitaula *et al*., 1995).

In two marshes, the vertical distribution of methane oxidation in the sediment and methane oxidation inhibition by ammonium was investigated by Van Der Nat *et al.* (1997). In a slurry incubation experiment, he conducted tests for two sites different in their prevailing vegetation type, i.e., reed and bulrush and in their heights above sea level. Inhibition of methane oxidation by ammonium was observed in all samples and depended on methane and ammonium concentrations. Increasing ammonium concentrations resulted in more inhibition and increasing methane concentrations resulted in less inhibition. Increasing atmospheric concentrations of CO_2 may increase emissions of N_2O by denitrification and either increase or decrease the ability of soil to buffer atmospheric CH_4 depending on fertilizer application (Baggs and Blum, 2004).

Effect of leachate recirculation, metals and minerals: Leachate recirculation is one option for inexpensive leachate disposal (Kinman *et al*., 1987; Cureton *et al*., 1991), in reducing the cost of post-closure care and long-term liability (Diamadopoulos, 1994; Westlake, 1995; Reinhart and Al-Yousfi 1996). It could participate to improve leachate quality; reduce volume of leachate to be treated and enhance gas production (Reinhart, 1996; Sulisti *et al*., 1996; Warith *et al*., 1999). Chan *et al.* (2002) found that leachate recirculation reduced waste stabilization time and was effective in enhancing gas production and improving leachate quality, especially in terms of Chemical Oxygen Demand (COD). The results also indicated that leachate recirculation could maximize the efficiency and waste volume reduction rate of landfill sites. Leachate recirculation gives an aqueous environment that assists the supply of nutrients and biomass within the landfill that stimulates the degradation of municipal solid waste (El-Fadel, 1999). Mobilizing nutrients and microorganisms in aqueous wastes improves mass transfer to prevent the development of inactive zones in landfill zones (Chugh *et al*., 1998). Leachate thus provides supply of COD, nutrients, biomass and water content with dissolved oxygen which stimulates methane oxidation.

Maurice *et al.* (1999) showed that larger trees arise on plots irrigated with leachate, presumably due to the positive effect of water and nutrient supply. Methane oxidation levels between 50 and 950 mol m^{-2} year were observed. The positive relationship between soil methane oxidation capacity and tree existence is important for reduction of methane emission by landfill vegetation type. Optimizing methane oxidation using vegetation on topsoil could reduce the amount of methane released to the atmosphere (Maurice *et al.*, 1999). Leachate provides the soils with higher content of water, nutrient and organic matter, it also provides metals. Heavy metals affect the growth, morphology and metabolism of microorganisms of soils through functional disturbance, protein denaturation, or the destruction of the integrity of the cell membrane (Baath, 1989; Babich *et al.*, 1980; Leita *et al.*, 1995). In a laboratory incubation study, Mishra *et al.* (1999) showed that selected heavy metals in three rice soils were different in their effect on methanogenesis and methane-producing bacteria. The Cd, Cu and Pb inhibited CH_4 production in all soils. The Zn stimulated CH_4 production in the alluvial soil, but inhibited it in laterite and acid sulfate soils. The Cr effectively inhibited CH_4 production in the alluvial soil, but stimulated it in laterite and acid sulfate soils (Mishra *et al.*, 1999).

In a laboratory study, Mohanty *et al.* (2000) showed that selected heavy metals differed in their effect on CH_4 oxidation in two soils for two water systems. The Cr significantly inhibited CH_4 oxidation in alluvial soil at 60% moisture capacity, while Cu stimulated the process. On the other hand, Zn inhibited CH_4 oxidation in both alluvial and laterite soils under saturated conditions.

The effect of inorganic redox substances (species of NO_3^-, Mn^{4+}, Fe^{3+} and SO_4^{-2}) on methane production and oxidation in anoxic rice soil samples has vital effects. Sulfate was the most inhibiting for methane production followed by Fe^{3+}, NO_3^- and Mn^{4+}, respectively. Laboratory studies showed that the addition of MnO_2 and K_2SO_4 enhanced aerobic methane oxidation in soil samples at 60% water content. Nitrate and Fe^{3+} motivated methane oxidation under anaerobic conditions and delayed it under aerobic conditions. Manganese (IV) delayed methane oxidation under anaerobic conditions, but enhanced it under aerobic conditions. On the other hand, SO_4^{-2} stimulated methane oxidation in soil equivalent medium under both aerobic and anaerobic conditions (Kumaraswamy *et al.*, 2001).

There is substantial interest in methane monooxygenase (MMOs) of methanotrophic bacteria in soils, because these enzymes in methanotrophs oxidize methane to a potential fuel source, methanol, detoxifies trichloroethylene and uses a greenhouse gas as a reactant. It is well known that sMMO expression in methanotrophs is repressed by copper ions sMMO is expressed for concentrations lower than 0.86 mmol g^{-1} dry cell weight (Barta and Hanson, 1993) or, generally, when the copper concentration is lower than 1 mM (Burrows *et al.*, 1984). Above this concentration, the particulate (membrane-bound) form of methane monooxygenase (pMMO) is produced (Nguyen *et al.*, 1994). To overcome this natural regulation, a *M. trichosporium* OB3b mutant has been obtained that expresses sMMO in the presence of copper probably because of a deficiency in copper transport (Phelps *et al.*, 1992). Transcription of soluble methane monooxygenase (sMMO) of methanotrophs is tightly regulated by low concentrations of copper ions [Cu^{2+} e.g., transcription is completely repressed at copper concentrations higher than 0.86 μmol g^{-1} dry cell weight] (Green *et al.*, 1985). In a research study, Jahng and Wood (1996) showed sMMO inhibition by metal ions and different medium ingredients was investigated for the first time using sMMO purified from the type II methanotroph *Methylosinus trichosporium* OB3b. Cu(I) and Cu(II) decreased sMMO activity of *Methylosinus trichosporium* OB3b by inhibiting not only the reductase but the hydroxylase component as well. Ni (II) also inhibited both enzyme components and Zn (II) inhibited sMMO by lowering the activity of the hydroxylase only. The Ni (II) and Zn (II) aggregated the reductase component of sMMO and the later precipitated the hydroxylase component. Cu (II) caused the reductase to precipitate (Jahng and Wood, 1996).

Uncertain environmental factors: Field studies in temperate forests have shown a chronological illustration of CO_2 and CH_4 fluxes (Castro *et al*., 1994, 1995; Peterjohn *et al*., 1994) that corresponds strongly with seasonal changes in soil moisture and temperature. Temperature is considered the primary predictor of CO_2 fluxes, not surprisingly; moisture also influences soil respiration rates (Groffman *et al*., 1992). Moisture usually exerts strong control over CH_4 uptake rates, although inclusion of both moisture and temperature in models can increase predictive capabilities. Lessard *et al*. (1994) suggested that the strong relationship between moisture and CH_4 uptake may mask relationships between temperature and uptake, thus it has been difficult to determine the relative importance of these factors. Steinkamp *et al*. (2001) showed significant seasonal differences in the magnitude of CH_4 oxidation rates at experimental sites with high rates during summer, relative low rates during winter and intermediate rates during spring and autumn. Hellebrand and Scholz (2000) showed results that the temperature was the main reason for the seasonal change of the methane uptake. Whereas the uptake dropped near to zero during the winter period, the uptake rates reached values up to 0.6 mg CH_4 m^{-2} day^{-1} (25 μg CH_4 m^{-2} h^{-1}) in the summer. Lessard *et al*. (1994) used field studies to determine the relative importance of moisture and temperature in controlling flux rates is difficult because soil temperature and moisture usually vary seasonally in temperate ecosystems. Soil temperatures are usually highest by late summer, but strong evapotranspiration potentials usually reduce soil water even if precipitation stays relatively constant. Thus, it is not a straightforward exercise to determine if maximum rates of soil respiration or CH_4 uptake in late summer, for example, are due to high temperatures, lower soil moisture, or an interaction of both factors.

BIOLOGICAL PLANT UPTAKE OF CARBON DIOXIDE

Measurements of CO_2 in the atmosphere, which began in Maunaloa (Hawaii) in 1958, indicated clearly that the concentration of CO_2 in the atmosphere is increasing rapidly (Keeling *et al*., 1982). The ice core studies showed that the CO_2 concentration was about 205 μmol mol^{-1} some 20,000 years ago. Pre-industrial value was 280 mmol mol^{-1} during the past 10,000 years. Whereas, before 1900, the CO_2 concentration in the atmosphere was 290 mmol mol^{-1}. Maunaloa studies also observed that from 1958-1982 there was an increase of 1.0 mmol mol^{-1} CO_2 per year. The 1958 value of atmospheric CO_2 was 316 mmol mol^{-1} and the present concentration were recorded as high as 370 mmol mol^{-1} (Kimball, 1997). Thus the concentration of CO_2 in the atmosphere is likely to be doubled (600 mmol mol^{-1}) by the middle of 21st century (Houghton *et al*., 1990). Recent studies suggest that boreal forests may play major role in regulating the climate of the northern hemisphere and in global carbon cycling (Bonan *et al*., 1992). In North America, the Boreal ecosystem atmosphere study program has investigated carbon and energy exchange in two regions of Canada (Sellers *et al*., 1995).

Short rotation woody crops fix carbon dioxide from the atmosphere and store carbon both above and belowground as biomass. Moreover, the harvested portions of the trees displace other products that are made from non-renewable fossil fuels (Tuskan and Walsh, 2001). One of the way to decrease greenhouse emissions in the future is to plant fast growing woody crops on unproductive land thereby sequestering carbon and displacing fossil fuels by harvesting woody biomass for bio-energy, or by storing carbon in long-lived woody products (Tuskan and Walsh, 2001). Short rotation poplar and willow crops deployed as phytoremediation buffer systems would qualify for carbon uptake and if planted at large scales would contribute greatly to atmospheric carbon dioxide gas reduction (Isebrands and Karnosky, 2001).

Examples of CO_2 plant uptake: Plants in their juvenile phase can benefit more than mature ones of optimal growing conditions. Transplant production in greenhouses offers the opportunity to optimize growing factors in order to reduce production time and improve transplant quality. Carbon dioxide and light are the two driving forces of photosynthesis. Carbon dioxide concentration can be enriched in the greenhouse atmosphere, leading to heavier transplants with thicker leaves and reduced transpiration rates (Tremblay and Gosselin, 1998). The almost entirely juvenile tissues of seedlings are all expanding and could be utilizing and diluting the enhanced photosynthate production in an enriched CO_2 atmosphere (Lindhout and Pet, 1990). Hence, the greatest advantage of CO_2 enrichment would be realized in the vegetative growth of young plants (Kimball, 1983). As leaf tissues formed early in seedling culture begin to mature, starch accumulation begins to slow photosynthetic rates and relative growth rate (Thomas *et al*., 1975).

The potential benefit from CO_2 augmentation of the greenhouse atmosphere has been known for a long time (Hand, 1984; Enoch, 1990) and they are particularly important for roses (Urban, 1994; Baille *et al*., 1996). The response of “Parfum de Rose” to CO_2 augmentation is rapid: it takes only 2 months to observe a sharp yield increase.

The net primary production of maturing loblolly pine (*Pinus taeda*), on a site of moderate soil fertility, increased by 25% in the two years following the onset of a 200 ppm CO_2 enrichment in a ‘Free Air CO_2 Enrichment’ FACE experiment (DeLucia *et al*., 1999). A parallel longer-term study produced a larger stimulation of tree growth of up to 34% over the first three years of enrichment (Oren *et al*., 2001). It is particularly interesting; therefore, that carbon allocation to reproduction is strongly stimulated in loblolly pine after three years of CO_2 enrichment (LaDeau and Clark, 2001). After this period, trees growing in the enriched CO_2 were twice as likely to be reproductively mature and produced three times as many cones and seeds, than control trees. In contrast, flowering and seed set in grasslands, where species may have deterministic life cycles, were stimulated (Smith *et al*., 2000).

C_4 and C_3 species increased total biomass significantly in elevated CO_2 by 33 and 44%, respectively and higher carbon assimilation rates were found in both C_3 and C_4 grass 33 and 25%, respectively (Wand *et al*., 1999). Carbon dioxide enrichment increased transplant leaf area, shoot and root dry weight and decreased the leaf area ratio of celery (*Apium graveolens* L.) transplants (Tremblay *et al*., 1987). Study on the characterization of CO_2 responsiveness in *Brassica oxycamp* hybrid and its parents *B. oxyrrhina* and *B. campestris* showed that the response of *B. oxycamp* hybrid to elevated CO_2 was significantly positive in respect to photosynthesis and growth, similar to that of its parent *B. campestris*. Hybrid *B. oxycamp* and *B. campestris* with greater sink potential responded significantly, whereas, *B. oxyrrhina* with poor sink size did not respond to CO_2 enrichment (Uprety *et al*., 1998).

Woodrow *et al*. (1987) demonstrated that CO_2 affects both source metabolism and partitioning to sinks (stems, roots and leaf carbohydrate) in tomato plantlets. They found that CO_2 enrichment produced heavier transplants desirable for successful field establishment without elongation growth. Dry matter accumulation in shoot and root was increased as well as leaf dry weight (by 81% over control). Transpiration rates were reduced under CO_2 enrichment conditions by 34%. Increased leaf dry weight accumulation and Specific Leaf Weight (SLW) under CO_2 enrichment suggests that more carbohydrate may be available to the plant for future growth. Apparently, the ratio of total sugars to amino acids in the leaf is shifted in favor of sugar content. In a study with tree seedlings, Luxmoore *et al*. (1986) suggest that CO_2 enrichment may increase sucrose translocation in roots and facilitate the mobilization of N and C compounds to new root primordia.

Increased net leaf photosynthesis rate and decreased transpiration rate under CO_2 enrichment are well documented (Woodrow *et al.*, 1987). One of the most important effects of CO_2 enrichment is the increase in water efficiency (Wong, 1979), which leads to drought tolerance. Actually, rising CO_2 concentration reduces the transpiration of plants by 20-40% (Mortensen, 1987). Radoglou *et al.* (1992) reported an increase in water use efficiency of bean (*Phaseolus vulgaris* L.) leaves as a result of increased assimilation rate and decreased stomatal conductance at higher ambient CO_2 concentrations. In cotton (*Gossypium arborcum* L.), stomata conductance after 40 day of CO_2 enrichment took 5 d to reach normal levels in non-enriched conditions. In short, CO_2 enrichment of vegetable transplants shortens the nursery period and modifies photosynthate allocation to the diverse parts, leading to sturdier, higher quality plants. This, together with the fact that CO_2 enriched plants make a more efficient use of water may impact favorably on the plant's ability to overcome transplanting stress (Sasek *et al.*, 1985). The relative increase in net assimilation rate due to an increase in CO_2 concentration from 200-1000 ppm was almost as great at the lower as at the higher light level studied. The light compensation point is lowered by increased CO_2 concentration (Mortensen, 1987). Fierro *et al.* (1994) demonstrated interactive effects of CO_2 and light enrichment on tomato and pepper transplants. If either were applied 3 weeks before transplanting tomatoes and peppers, they increased accumulation of dry matter in shoots by 50%. Fierro *et al.* (1994) results suggested that it is more important to achieve optimal light conditions first and then make use of CO_2 enrichment.

CO_2 plant uptake mechanism: Plants and other photosynthetic organisms convert carbon dioxide into energy rich organic molecules, which are the precursors of carbohydrates. These molecules are either used by the organism directly or are passed into the food chain to be utilized by non-photosynthetic organisms (Paradise and Cyr, 1995). The complete photosynthetic process may be summarized by the following equation:

$$nCO_2+nH_2O\rightarrow(CH_2O)n+nO_2$$

Carbon dioxide serves as substrate to photosynthetic carbon assimilation. There is concomitant decline in photo-respiratory activity and alteration in stomatal aperture. It was reported that C_3 plants (wheat, rice, oilseeds, pulses, etc.) respond to elevated CO_2 since elevated CO_2 reduces oxygenase activity of RuBP carboxylase oxygenase enzyme in plants. The C_4 plants (sorghum, maize, sugarcane, etc.) show little or no photosynthetic response to elevated CO_2 because C_4 pathway is not competitively inhibited by O_2 and is completely CO_2 saturated. However, there is no consensus on the quantitative effects of increased CO_2 in plant processes and growth due to differences in response at different stages of growth, species of crops and because of growth limiting environmental factors (Uprety *et al.*, 2000a).

The key enzyme for CO_2 fixation is rubisco. Its activity depends on the ratio of the O_2 and CO_2 concentration in the atmosphere. The major effect of CO_2 enrichment is the shift in balance between the carboxylation and oxygenation activity of rubisco, this effect is important at low and high light levels. Kimball (1983) stated that, on average, yields of crops should increase by 33% with a doubling of CO_2 concentration in the earth's atmosphere. Although these estimates have been developed for plants over their complete life cycles, enhanced growth and dry matter accumulation are correlated with higher net photosynthetic rates in young vegetative tissues under CO_2 enrichment as well.

Studies on the response of rice cultivars Basmati-1 and Pusa-677 to the elevated CO_2 showed increase in growth, photosynthesis and seed yield (Uprety *et al.*, 2000b). The increased photosynthesis and greater accumulation of sugar contributed significantly to the accelerated development of leaves and tillers in both cultivars. The response of rice cultivars to the elevated CO_2 with reference to their stomatal characters showed a marked increase in the stomatal resistance, stomatal index, size of guard cells, stroma and epidermal cells (Uprety *et al.*, 2002).

Parameters affecting CO_2 plant uptake: Ecosystem responses to CO_2 enrichment and climate change are expressed at different levels: biochemical and stomatal processes at the leaf level, growth processes at the plant level and water budget and carbon-nitrogen cycling at the ecosystem level. Predicted responses of net primary production and N mineralization to CO_2 enrichment and climate change are, therefore, complex. Responses to individual climate factors were often modified by responses to other factors, through interaction among processes at the same and different levels of organization (Coughenour and Chen, 1997). The final outcome of ecosystem responses to CO_2 enrichment and climate change is dependent on the relative importance of these different processes. The relative enhancement of net primary production by CO_2 enrichment was greater at high temperature in the C_3 grass species and greater under ambient temperature conditions in the C_4 grass. Most previous studies revealed that a CO_2-doubling generally caused greater increases in growth at higher temperatures (Cure, 1985). Although, Sionit *et al.* (1981) reported that the CO_2 response of okra had stimulatory effects on growth at lower temperature and Coleman and Bazzaz (1992) reported that the effects of CO_2 enrichment on final biomass of a C_4 annual forb was positive at low temperature and negative at high temperature. Chen *et al.* (1994) found that the relative enhancement of CO_2 enrichment on an aquatic plant was strongly temperature dependent and that the maximum relative effects of CO_2 were reached at a low temperature. These different responses suggest that the effects of elevated CO_2 and temperature may interact in a seasonal environment and that this interaction is species specific (Hunt *et al.*, 1991).

The responses of N mineralization to elevated CO_2 were negative or near neutral except for the C_3 grass species system at high precipitation. Reduced N mineralization under elevated CO_2 may be ascribed to reduced litter quality (Strain and Bazzaz, 1983). However, N mineralization in the *P. smithii* system was increased by doubling CO_2 under wet conditions. Increased N mineralization under elevated CO_2 was also observed in an open top hamper study of *Populus granditenta* (Zak *et al.*, 1993). They found up to five-fold increases in N mineralization. They suggested that elevated CO_2 would increase below ground carbon inputs, which would increase microbial biomass and their activity and thus decomposition and mineralization rate. Decomposition and mineralization could be greater under elevated CO_2 in spite of decreased residue quality, because decomposition and mineralization, being donor-controlled processes, tend to increase with net primary production (Hunt *et al.*, 1991; Coughenour and Chen, 1997). Interactions among biochemical, ecophysiological and ecosystem processes appear to be significant. To reasonably estimate ecological effects of CO_2 enrichment and climate change, a model must couple biochemical, ecophysiological and ecosystem processes. Ecosystem models that ignore physiological and ecophysiological responses are incapable of predicting responses to elevated CO_2 and climate change (Coughenour and Chen, 1997). On the other hand, ecophysiological models that do not couple soil water and nitrogen cycling would be not able to represent the feedbacks of biogeochemical processes such as N mineralization. In natural ecosystems, many plant species coexist competitively. There are hundreds of plant species representing C_3, C_4 and CAM photosynthetic pathways in Colorado shortgrass steppe. The CO_2 enrichment and climate change differently affect many plant and ecosystem processes. It has been hypothesized that different responses of C_3 and C_4 species to CO_2 enrichment and climate change could change the competitive balance between these species in a future CO_2 enriched atmosphere (Arp *et al*., 1993).

Trees treated with both nitrogen fertilizer and elevated CO_2 showed a three-fold increase in growth over controls. The photosynthetic rates of plant species within a grassland community also showed little positive response to CO_2 enrichment (Lee *et al*., 2001) observed after at least three years of fumigation (Oren *et al*., 2001). The stimulation of carbon sequestration in the vegetation and litter of forest trees was paralleled in grassland where CO_2 was enriched by 240 ppm over a six-year period (Niklaus *et al*., 2001). The elevated CO_2 increased the rate of photosynthesis, promoted greater foliage, more number of siliqua and increased root growth. The carbon need to satisfy the demand for these newly generated sinks was met from new photosynthates produced during the high CO_2 exposure. Plants accumulated increased metabolic reserves in terms of sugars and starch to significantly enhance metabolic flexibility to respond to adverse moisture stress (Uprety *et al*., 1995). The main constraint to growth occurs when the plant is unable to photosynthesize at the potential rate, governed by the amount of incoming solar radiation. This will normally occur when one of the substrates for photosynthesis (carbon, water, nitrogen) is limiting (Agren, 1985).

Plant uptake of CO_2 in landfill: The landfill industry makes use of a potentially valuable, environmentally beneficial and sustainable utilization of restored landfill sites. The biomass produced at restored landfill sites from Short Rotation Coppice (SRC) trees, such as willow and poplar, may have a number of uses: as a fuel for electricity generation plants; for the production of charcoal; as a soil amendment for clay caps; or simply as a carbon sink for atmospheric CO_2 (Cannell *et al*., 1987).

Present municipal solid waste landfills generate biogas that is flared on site to destroy noxious contaminants and water is extracted from leachate to be drained away. However, biogas could alternatively be a cheap fuel for winter heating and could provide horticultural greenhouses with abundant carbon dioxide to boost plant growth all year long (Jaffrin *et al*., 2003).

In the research of Chan *et al*. (1997), two landfill sites in Hong Kong (Shuen Wan and Junk Bay StageI) were chosen as the field sites and three sites (Yuen Chau Kok, Tai Po and Lok Wo Sha) were chosen as reference sites. The vegetation survey classified the plants under woody plants, herbs and grasses groups. Seventeen species of vascular plants were found at the reference sites; whereas 25 and 20 species, respectively, were found at the two-landfill sites. The gross average coverage of plants at the reference sites (57.8%) was lower than that at the two-landfill sites (70.3, 61.4%). Tree cover on the landfill sites was dominated by two species: *Acacia confusa* and *Leucaena leucocephala,* which seemed suited for growth and were abundant on landfill sites. Therefore, completed landfill sites can be good habitats, which support a variety of plants and animals, provided that the plants can phytoremediate landfill gas and leachate contamination (Chan *et al*., 1997).

PHYSICAL BIOGAS EXTRACTION

Biogas collection systems: In landfills, gas is collected using network of collection pipes and wells. The gas collection efficiency in landfills is between 40-90% (Augenstein and Pacey, 1991). Biogas is collected by means of some vertical and horizontal drain pipes and is then burned in flares or employed to produce heat and energy (EMCON Associates, 1980). The designs for gas abstraction systems include different types of well configurations such as vertical, horizontal, hybrid, etc (Leach, 1990). The landfill gas collection system consists of vertical extraction well, transport pipe network, blower for passive gas collection or pumping for active collection, monitoring equipment and flare stack (Park and Shin, 2001).

Landfill gas can be collected by either a passive or an active collection system. Passive gas collection systems (Fig. 1) use existing variations in landfill pressure and gas concentrations to vent landfill gas into the atmosphere or a control system. The collection wells are typically constructed of perforated or slotted plastic and are installed vertically throughout the landfill to depths ranging from 50-90% of the waste thickness. Vertical wells are typically installed after the landfill or a portion of a landfill, has been closed. A passive collection system may also include horizontal wells located below the ground surface to serve as conduits for gas movement within the landfill (ATSDR., 2001).

Active gas collection (Fig. 2) is considered a good means of landfill gas collection (EPA., 1991). An active collection system composed of extraction wells connected to header pipe to a pump that

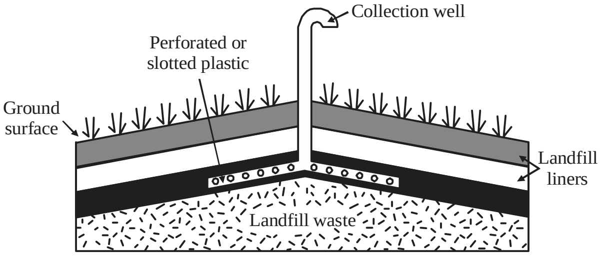

Fig. 1: Passive gas collection system (ATSDR., 2001)

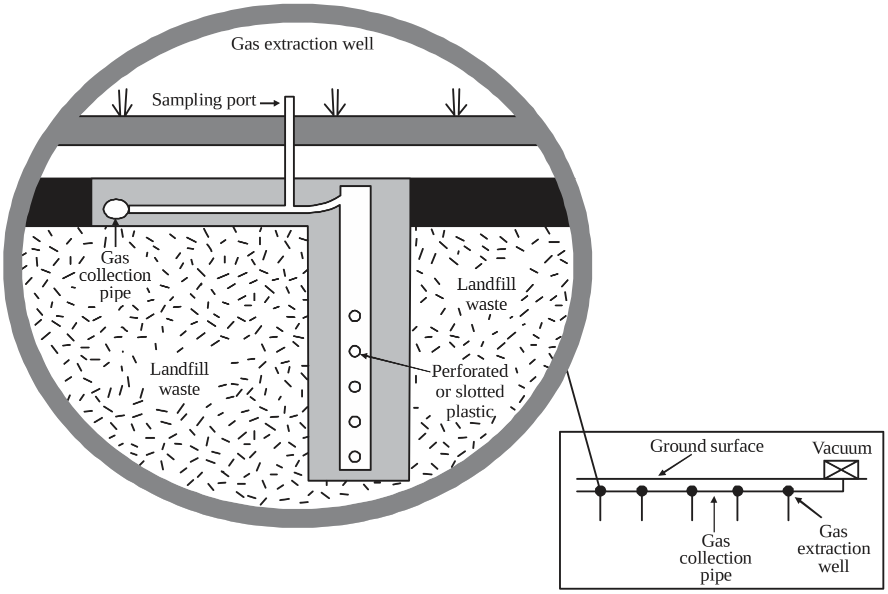

Fig. 2: Active gas collection system (ATSDR., 2001)

delivers gas for energy recovery (Wheless and Wiltsee, 2001). An active system consists of extraction well, header pipe, vacuum, condensate removal and burner (Bagchi, 1994, 2004). However, wells in the active system should have valves to regulate gas flow and to serve as a sampling port. Sampling allows the system operator to measure gas generation, composition and pressure (ATSDR., 2001).

The Environmental Protection Agency (EPA), state and local regulators use assumed gas collection efficiencies to calculate landfill emissions in regulating and setting policies for landfills. These assumed efficiencies are usually around 75%. A thorough review of gas system operators' experiences and of researcher's efficiency determination methods and their results is required to better assess actual collection efficiencies. As stated by Sanitation Districts of Los Angeles County in 2015.

Biogas control methods: There are two common methods of conducting a control to gas migration: using very low permeability materials to resist gas flow and using highly permeable materials to allow the gas to vent to the surface. Among combination of those methods the following could be summarized:

- Trench filled with granular backfill acts as passive system to collect gas from boundaries
- Trench backfilled with gravel and an impermeable membrane installed along the back wall of the trench to trap the gas to be more effective interceptor
- A gravel trench with vertical perforated pipe to collect gas within trench passive collection system
- A gravel trench with horizontal and vertical perforated pipes to collect gas within the trench (McBean *et al*., 1995)

Current methods of forming a gas resistant barrier usually involve the excavation of a trench and backfilling with either a low permeable material such as bentonite or the inclusion of a gas resistant membrane. Vent trenches are normally constructed using trenches backfilled with either gravel or geocomposite venting media to promote gas flow to the surface. An alternative method is to provide a series of discrete vent wells at regular spacing or using vertical permeable curtain. These methods allow the gas to exhaust directly to atmosphere without any dilution in the system (Wilson and Shuttleworth, 2002). The Vent System provide preferential pathway (route of least resistance) to atmosphere as shown in Fig. 3 (Permavoid Ltd., 2015).

Biogas collection layers: There are other processes of gas collection such as collection layers. Tire shreds have been used as landfill gas collection material at municipal solid waste landfills (Augustein *et al*., 1997). GeoSyntec Consultants (1998a) for the California Integrated Waste Management Board provided a summary of recommended procedures for use of tire shreds as landfill gas collection material at municipal solid waste landfills, including horizontal landfill gas collection layers, horizontal trenches and vertical boreholes. Tire shreds have a compressibility that is several orders of magnitude greater than materials typically used for landfill gas collection such as gravel, sand, or crushed rock (GeoSyntec Consultants, 1998b). Consideration should be given to using a properly selected geotextile as a separation between the tire shreds and soil materials. The geotextile would prevent soil from migrating into the relatively large voids between the tire

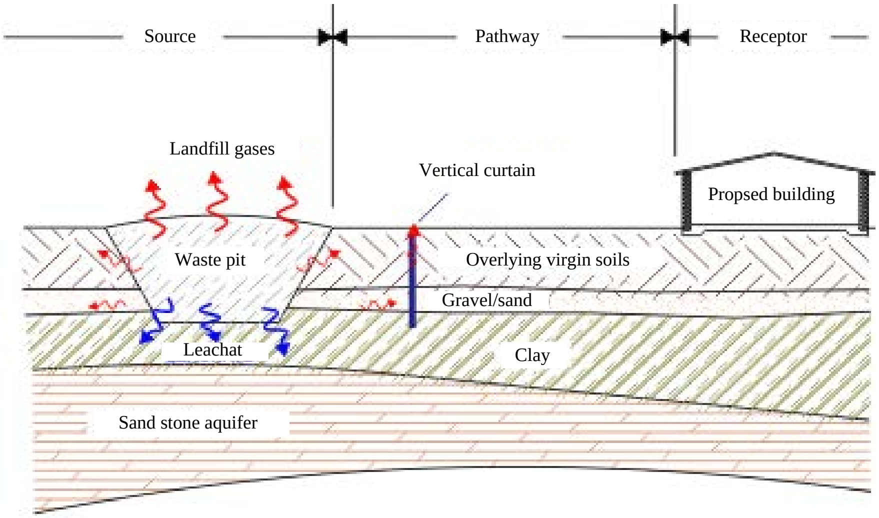

Fig. 3: Vertical permeable venting curtain for gas control

shreds (GeoSyntec Consultants, 1998a). According to intrinsic permeability values, it is found that grain size fractions of 0.1-2, 0.6-1.2 (sand) and 0.1-56 mm (natural soil), are inappropriate for the application as gas collection layer materials, as they demonstrated lower values than a sample of solid waste (Huber and Wohnlich, 1999). As the mentioned grain size fractions of sand and natural soil belong to materials, which are commonly used for constructing the foundation layer of landfill cover systems (Fox, 1996), these layers cannot serve at the same time as gas collection layers. The coarse materials (grain size fraction of 16-32 mm of gravel, crushed lime stone, crushed granite and crushed basalt) were proved to be the most suitable for constructing gas collection layers, since it indicates high intrinsic permeability (Huber and Wohnlich, 1999).

Biogas collection covers: There are other methods of gas collection such as collection covers, which are used other than application in landfill, but could give good analogy of incorporated processes and/or materials to apply in landfill.

McGrath and Mason (2004) in their application to assess biogas production from anaerobic waste stabilization pond treating farm dairy wastewater, they used floating gas collection cover was constructed of fiber-reinforced polypropylene geomembrane fabric (J.P.S. Elastomerics, Westfield, NC, USA), supported on a 2 m by 2 m framework fabricated from 25 mm diameter and foam filled 110 mm diameter polyvinylchloride pipe and fittings.

Manure lagoons can be covered by floating polyethylene liners on the liquid surface allowing for the cost-effective construction of anaerobic digesters which use microorganisms to ferment and process waste producing methane gas. The polyethylene cover prevents vector migration out of the lagoon preventing odors, vapors and gasses from entering the atmosphere. They also prevent rainwater dilution. The most desirable effect of the polyethylene cover is to create an inflatable methane gas reservoir (Kriofske, 1998).

The XR-5 cover by ETP Inc, AL USA, is a proven material for biogas collection. It is strong, tough and it is highly resistant to biogas. Its low gas permeability makes it ideal for the collection of biogas and for use as a floating cover material. Poly-log-floats are used to buoy the cover and to form pass-ways for the collection of the biogas. Rainwater drainage is provided by the use of thru-cover drains. The cover's unique design allows it to act as a gas storage container at times of peak production (ETP Inc., 2015).

A large employer in upper State Maine, USA, expanded their food processing capability. As part of that expansion, they needed to build an anaerobic digester that would retain heat in the cold climate of Maine and be able to collect the biogas produced. Lemna, MN, USA used LemTec™ Modular Insulated Cover System a turnkey to biogas design (Lemna Technologies Inc., 2015).

Geomembrane Technologies Inc. GTI, NB, Canada produces gas collection covers that are floating cover systems used to collect gases from wastewater treatment lagoons, sludge ponds, aeration systems, flow equalization tanks and pretreatment tanks. In the case of anaerobic digestors, GTI cover systems are used to capture biogas. The GTI cover system includes reduction of process heat loss/gain, elimination of water evaporation and prevention of sunlight penetration. Features can include insulation, automatic rainwater removal, baffles, sample ports and hatches. These durable, UV protected covers are strong enough to safely support foot traffic, light vehicles and snow loads (GTI., 2015).

The Canadian french-fry manufacturer, needed to replace an old (competitor's) insulated gas collection cover that was prone to failure due to migrating solids (grease bergs) that ripped the gas collection membrane. The design of the old cover made it next to impossible to remove the grease bergs and to repair the damaged cover. The insulated gas collection cover by Lemna replaced the old one. The new gas collection cover has the ability to remove sections of the cover to dispel solids when required (Lemna Technologies Inc., 2015).

A dairy in Pennsylvania had a wastewater tank that was generating odors. Lemna Technologies proposed the installation of LemTec™ Gas Collection Cover System. This system provides effective odor control by completely covering the water surface with a single layer membrane. Channels are created beneath the cover for gas to flow for collection. Channels above the cover provide for rainwater removal (Lemna Technologies Inc., 2015).

Scotford and Williams (2001) investigated the effectiveness of a floating plastic cover to reduce ammonia emissions from slurry lagoon. They measured ammonia emissions from both an uncovered lagoon and covered lagoon. The ammonia emissions were measured using hoods. Each hood was fitted with an inlet pipe and outlet pipe. Air was forced into the hood using a fan that gave a superficial air velocity across the measured surface. The ammonia flux was measured into and out of the hoods. The floating plastic cover prevented nearly 100% of these emissions.

CONCLUSION

Biogas in landfill is intercepted by different means of biological and physical processes. The biological processes are natural activities such as bacterial methane oxidation and plant uptake for carbon dioxide at topsoil layer. Methane oxidation is affected by different environmental factors such as: temperature, water content, nutrients, substrate and oxygen concentrations. Plant uptake for carbon dioxide is affected by some parameters such as: CO_2 concentration, nitrogen concentration, water content and temperature. The physical engineered processes are represented by various physical biogas extraction systems. Landfill gas can be collected by either a passive or an active collection system. Passive gas collection systems depend on variations of gas pressures and concentrations. An active collection system is an extraction systems depend on pumping that delivers gas for energy use.

REFERENCES

ATSDR., 2001. Landfill gas primer-an overview for environmental health professional. US. Department of Health and Human Services, Atlanta, GA., http://www.atsdr.cdc.gov/hac/landfill/html/ch1.html.

Abichou, T., K. Mathieu, L. Yuan, J. Chanton and G. Hater, 2009. Effects of compost biocovers on gas flow and methane oxidation in a landfill cover. Waste Manage., 29: 1595-1601.

Adams, R.S. and R. Ellis, 1960. Some physical and chemical changes in the soil brought about by saturation with natural gas. Soil Sci. Soc. Am. J., 24: 41-44.

Agren, G.I., 1985. Limits to plant production. J. Theor. Biol., 113: 92-98.

Andreottola, G. and R. Cossu, 1988. Modello matematico di produzione del biogas in uno scarico controllato [Mathematical model of biogas production]. Rifituti Solidi, 2: 473-483.

Anthony, C., 1986. Bacterial oxidation of methane and methanol. Adv. Microb. Physiol., 27: 113-210.

Arp, W.J., B.G. Drake, W.T. Pockman, P.S. Curtis and D.F. Whigham, 1993. Interactions between C_3 and C_4 salt marsh plant species during four years of exposure to elevated atmospheric CO_2. Adv. Vegetation Sci., 14: 133-143.

Augenstein, D. and J. Pacey, 1991. Modelling landfill methane generation. Proceedings of Sardinia 91, 3rd International Landfill Symposium, October 14-18, 1991, CISA., Cagliari, Italy, pp: 115-148.

Augustein, D., R. Yazdani, R. Moore and K. Dahl, 1997. Yolo county controlled landfill demonstration project. Proceedings of the SWANA's 2nd Annual Landfill Symposium, August 4-6, 1997, Sacramento, California, pp: 3-39.

Baath, E., 1989. Effects of heavy metals in soil on microbial processes and populations: A review. Water Air Soil Pollut., 47: 335-379.

Babich, H., G. Stotzky and H.L. Ehrlich, 1980. Environmental factors that influence the toxicity of heavy metal and gaseous pollutants to microorganisms. CRC Crit. Rev. Microbiol., 8: 99-145.

Bagchi, A., 1994. Design, Construction and Monitoring of Landfills. 2nd Edn., John Wiley and Sons Inc., New York, Pages: 361.

Bagchi, A., 2004. Design of Landfills and Integrated Solid Waste Management. 3rd Edn., John Wiley and Sons Inc., New Jersey, USA., ISBN-13: 978-0471254997, Pages: 712.

Baggs, E.M. and H. Blum, 2004. CH_4 oxidation and emissions of CH_4 and N_2O from *Lolium perenne* swards under elevated atmospheric CO_2. Soil Biol. Biochem., 36: 713-723.

Bailey, J.E. and D.F. Ollis, 1986. Applied Enzyme Catalysis, in Biochemical Engineering Fundamentals. 2nd edn, McGraw-Hill, New York, ISBN-13: 9780070032125, Pages: 984.

Baille, M., R. Romero-Aranda and A. Baille, 1996. Gas-exchange responses of rose plants to CO_2 enrichment and light. J. Hortic. Sci. Biotechnol., 71: 945-956.

Barta, T.M. and R.S. Hanson, 1993. Genetics of methane and methanol oxidation in gram-negative methylotrophic bacteria. Antonie Van Leeuwenhoek, 64: 109-120.

Bender, M. and R. Conrad, 1992. Kinetics of CH_4 oxidation in oxic soils exposed to ambient air or high CH_4 mixing ratios. FEMS Microbiol. Lett., 101: 261-269.

Bender, M. and R. Conrad, 1994. Microbial oxidation of methane, ammonium and carbon monoxide and turnover of nitrous oxide and nitric oxide in soils. Biogeochemistry, 27: 97-112.

Boeckx, P. and O. van Cleemput, 1996. Methane oxidation in a neutral landfill cover soil: Influence of moisture content, temperature and nitrogen-turnover. J. Environ. Qual., 25: 178-183.

Boeckx, P., O. van Cleemput and I. Villaralvo, 1996. Methane emission from a landfill and the methane oxidising capacity of its covering soil. Soil Biol. Biochem., 28: 1397-1405.

Bogner, J., K. Spokas, E. Burton, R. Sweeney and V. Corona, 1995. Landfills as atmospheric methane sources and sinks. Chemosphere, 31: 4119-4130.

Bogner, J.E., K.A. Spokas and E.A. Burton, 1997. Kinetics of methane oxidation in a landfill cover soil: Temporal variations, a whole-landfill oxidation experiment and modeling of net CH_4 emissions. Environ. Sci. Technol., 31: 2504-2514.

Bonan, G.B., D. Pollard and S.L. Thompson, 1992. Effects of boreal forest vegetation on global climate. Nature, 359: 716-718.

Borjesson, G. and B.H. Svensson, 1997. Seasonal and diurnal methane emissions from a landfill and their regulation by methane oxidation. Waste Manage. Res., 15: 33-54.

Bowden, R.D., K.M. Newkirk and G.M. Rullo, 1998. Carbon dioxide and methane fluxes by a forest soil under laboratory-controlled moisture and temperature conditions. Soil Biol. Biochem., 30: 1591-1597.

Burrows, K.J., A. Cornish, D. Scott and I.J. Higgins, 1984. Substrate specificities of the soluble and particulate methane mono-oxygenases of methylosinus trichosporium OB3b. J. Gen. Microbiol., 130: 3327-3333.

Cannell, M.G.R., R. Milne, L.J. Sheppard and M.H. Unsworth, 1987. Radiation interception and productivity of willow. J. Applied Ecol., 24: 261-278.

Castro, M.S., J.M. Melillo, P.A. Steudler and J.W. Chapman, 1994. Soil moisture as a predictor of methane uptake by temperate forest soils. Can. J. For. Res., 24: 1805-1810.

Castro, M.S., P.A. Steudler, J.M. Melillo, J.D. Aber and R.D. Bowden, 1995. Factors controlling atmospheric methane consumption by temperate forest soils. Global Biogeochem. Cycles, 9: 1-10.

Chalvatzaki, E. and M. Lazaridis, 2009. Assessment of air pollutant emissions from the Akrotiri landfill site (Chania, Greece). Waste Manage. Res., 28: 778-788.

Chan, G.Y.S., L.M. Chu and M.H. Wong, 1997. Influence of landfill factors on plants and soil fauna-an ecological perspective. Environ. Pollut., 97: 39-44.

Chan, G.Y.S., L.M. Chu and M.H. Wong, 2002. Effects of leachate recirculation on biogas production from landfill co-disposal of municipal solid waste, sewage sludge and marine sediment. Environ. Pollut., 118: 393-399.

Chanakya, H.N., I. Sharma and T.V. Ramachandra, 2009. Micro-scale anaerobic digestion of point source components of organic fraction of municipal solid waste. Waste Manage., 29: 1306-1312.

Chanton, J., T. Abichou, C. Langford, K. Spokas and G. Hater *et al*., 2011. Observations on the methane oxidation capacity of landfill soils. Waste Manage., 31: 914-925.

Chen, D.X., M.B. Coughenour, D. Eberts and J.S. Thullen, 1994. Interactive effects of CO_2 enrichment and temperature on the growth of dioecious *Hydrilla verticillata*. Environ. Exp. Bot., 34: 345-353.

Christensen, T.H., P.A. Kjeldsen and B. Lindhardt, 1996. Gas Generating Processes in Landfills. In: Landfilling of Waste: Biogas, Christensen, T.H., R. Cossu and R. Stegman (Eds.). E and FN Spon, London, UK., pp: 25-50.

Christophersen, M., L. Linderod, P.E. Jensen and P. Kjeldsen, 2000. Methane oxidation at low temperatures in soil exposed to landfill gas. J. Environ. Qual., 29: 1989-1997.

Chugh, S., W. Clarke, P. Pullammanappallil and V. Rudolph, 1998. Effect of recirculated leachate volume on MSW degradation. Waste Manage. Res., 16: 564-573.

Coleman, J.S. and F.A. Bazzaz, 1992. The effects of CO_2 and temperature on growth and resource use of co-occurring C_3 and C_4 annuals. Ecology, 73: 1244-1259.

Coughenour, M.B. and D.X. Chen, 1997. Assessment of grassland ecosystem responses to atmospheric change using linked plant-soil process models. Ecol. Applic., 7: 802-827.

Cure, J.D., 1985. Carbon Dioxide Doubling Responses: A Crop Survey. In: Direct Effects of Increasing Carbon Dioxide on Vegetation, Strain, B.R. and J.D. Cure (Eds.). U.S. Department of Energy, Washington, DC., pp: 99-116.

Cureton, P.M., P.H. Groenevelt and R.A. McBride, 1991. Landfill leachate recirculation: Effects on vegetation vigor and clay surface cover infiltration. J. Environ. Qual., 20: 17-24.

Czepiel, P.M., B. Mosher, P.M. Crill and R.C. Harris, 1996. Quantifying the effect of oxidation on landfill methane emissions. J. Geophys., 101: 16721-16729.

Davis, M.L. and D.A. Cornwell, 2008. Introduction to Environmental Engineering. 4th Edn., McGraw-Hill, New York, USA., ISBN-13: 9780071259224, Pages: 1008.

DeLucia, E.H., J.G. Hamilton, S.L. Naidu, R.B. Thomas and J.A. Andrews *et al*., 1999. Net primary production of a forest ecosystem with experimental CO_2 enrichment. Science, 284: 1177-1179.

Diamadopoulos, E., 1994. Characterization and treatment of recirculation-stabilized leachate. Water Res., 28: 2439-2445.

Dunfield, P., R. Knowles, R. Dumont and T.R. Moore, 1993. Methane production and consumption in temperate and subarctic peat soils: response to temperature and pH. Soil Biol. Biochem., 25: 321-326.

Duxbury, J.M., 1994. The significance of agricultural sources of greenhouse gases. Fertilizer Res., 38: 151-163.

EMCON Associates, 1980. Methane Generation and Recovery from Landfills. Ann Arbor Science Publishers, Ann Arbor, MI., USA.

EPA., 1991. Air emissions from municipal solid waste landfills: Background information for proposed standards and guidelines. EPA-450/3-90/011a, March 1991, U. S. Environmental Protection Agency, USA.

ETP Inc., 2015. Biogas collection covers. Engineered Textile Products Inc., USA. http://www.etpinfo.com/biogas.htm.

El-Fadel, M., 1991. Modeling gas and heat generation and transport in sanitary landfills. Ph.D. Thesis, Stanford University, Stanford, CA.

El-Fadel, M., 1999. Leachate recirculation effects on settlement and biodegradation rates in MSW landfills. Environ. Technol., 20: 121-133.

Enoch, H., 1990. Crop responses to aerial carbon dioxide-refereed paper. Acta Hortic., 268: 17-32.

Fierro, A., A. Gosselin and N. Tremblay, 1994. Supplemental carbon dioxide and light improved tomato and pepper seedling growth and yield. HortScience, 29: 152-154.

Fox, P.J., 1996. Landfill Cover Systems in the United States. In: Alternative Dichtungsmaterialien im Deponiebau und in der Altlastensicherung-Innovative, kostergunstige und gleichwertige Losungen, Burkhardt, G. and T. Egloffstein (Eds.). University of Karlsruhe, Germany, pp: 1-24.

GTI., 2015. Biogas energy recovery. Geomembrane Technologies Inc., USA. http://www.gticovers.com/ applications/biogas-energy-recovery.

Galloway, J.N., W.H. Schlesinger, H. Levy II, A. Michaels and J.L. Schnoor, 1995. Nitrogen fixation: Anthropogenic enhancement-environmental response. Global Biogeochem. Cycles, 9: 235-252.

GeoSyntec Consultants, 1998a. Guidance manual-tire shreds as gas collection material at municipal solid waste landfills. GeoSyntec Consultants Inc., California.

GeoSyntec Consultants, 1998b. Test pad demonstration program-tire shreds as cover foundation, leachate drainage and operations layer material at municipal solid waste landfills. GeoSyntec Consultants Inc., California.

Green, J., S.D. Prior and H. Dalton, 1985. Copper ions as inhibitors of protein c of soluble methane monooxygenase of *Methylococcus capsulatus* (Bath). Eur. J. Biochem., 153: 137-144.

Groffman, P.M., A.J. Gold and R.C. Simmons, 1992. Nitrate dynamics in riparian forests: Microbial studies. J. Environ. Qual., 21: 666-671.

Hand, D.W., 1984. Crop responses to winter and summer CO_2 enrichment. Acta Hortic., 162: 45-64.

Hanson, R.S. and T.E. Hanson, 1996. Methanotrophic bacteria. Microbiol. Rev., 60: 439-471.

Hellebrand, H.J. and V. Scholz, 2000. Influence of plants and fertilisation level on the microbial uptake of methane in soils. Proceedings of the 14th Memorial CIGR World Congress, November 28-December 1, 2000, Tsukuba, Japan.

Houghton, J.T., G.J. Jenkins and J.J. Ephraumas, 1990. Climate Change - The PCC Scientific Assessment. Cambridge University Press, Cambridge.

Houghton, J.T., L.G.M. Filho, B.A. Callandar, N. Harriss, A. Kattenberg and K. Maskell, 1996. Technical Summary. In: Climate Change 1995: The Science of Climate Change, Houghton, J.T., L.G.M. Filho, B.A. Callandar, N. Harriss, A. Kattenberg and K. Maskell (Eds.). Cambridge University Press, Cambridge, ISBN: 9780521564366, pp: 9-49.

Huber, A. and S. Wohnlich, 1999. Gas collection layers. Proceedings Sardinia 99, Seventh international waste management and landfill symposium, October 4-8, 1999, S. Margherita di Pula, Cagliari, Italy.

Hunt, H.W., M.J. Trlica, E.F. Redent, J.C. Moore and J.K. Detling *et al*., 1991. Simulation model for the effects of climate change on temperate grassland ecosystems. Ecol. Modell., 53: 205-246.

Hutsch, B.W., 1998. Methane oxidation in arable soil as inhibited by ammonium, nitrite and organic manure with respect to soil pH. Biol. Fertil. Soils, 28: 27-35.

Im, J., S. Moon, K. Nam, Y.J. Kim and J.Y. Kim, 2009. Estimation of mass transport parameters of gases for quantifying CH_4 oxidation in landfill soil covers. Waste Manag., 29: 869-875.

Ionescu, G., E.C. Rada, M. Ragazzi, C. Marculescu, A. Badea and T. Apostol, 2013. Integrated municipal solid waste scenario model using advanced pretreatment and waste to energy processes. Energy Convers. Manage., 76: 1083-1092.

Isci, A. and G.N. Demirer, 2007. Biogas production potential from cotton wastes. Renewable Energy, 32: 750-757.

Isebrands, J.G. and D.F. Karnosky, 2001. Environmental Benefits of Poplar Culture. In: Poplar Culture in North America, D.I. Dickmann, J.G. Isebrands, J.E. Eckenwalder and J. Richaedson (Eds.). NRC Research Press, Ottawa, Ontario, Canada, pp: 207-218.

Jaffrin, A., N. Bentounes, A.M. Joan and S. Makhlouf, 2003. Landfill biogas for heating greenhouses and providing carbon dioxide supplement for plant growth. Biosyst. Eng., 86: 113-123.

Jahng, D. and T.K. Wood, 1996. Metal ions and chloramphenicol inhibition of soluble methane monooxygenase from *Methylosinus trichosporium* OB3b. Applied Microbiol. Biotechnol., 45: 744-749.

Jones, H.A. and D.B. Nedwell, 1993. Methane emission and methane oxidation in land-fill cover soil. FEMS Microbiol. Ecol., 11: 185-195.

Keeling, C.D., R.B. Bacastow and T.P. Whorf, 1982. Measurements of the Concentration of Carbon Dioxide at Maunaloa Observatory, Hawaii. In: Carbon Dioxide Review, Clark, W.C. (Ed.)., Oxford University Press, New York pp: 377-385.

Kightley, D., D.B. Nedwell and M. Cooper, 1995. Capacity for methane oxidation in landfill cover soils measured in laboratory-scale soil microcosms. Applied Environ. Microbiol., 61: 592-601.

Kimball, B.A., 1983. Carbon dioxide and agricultural yield: An assemblage and analysis of 430 prior observations. Agron. J., 75: 779-788.

Kimball, B.A., 1997. Influence of increasing CO_2 concentration on photosynthetic stimulation of selected weeds. Photosynth. Res., 54: 199-208.

King, G.M., 1992. Ecological Aspects of Methane Oxidation, a Key Determinant of Global Methane Dynamics. In: Advances in Microbial Ecology, Marshall, K.C. (Ed.)., Plenum Press, New York, pp: 431-468.

King, M.A. and A.P.S. Adamsen, 1992. Effects of temperature on methane consumption in a forest soil and in pure cultures of the methanotroph *Methylomonas rubra*. Applied Environ. Microbiol., 58: 2758-2763.

King, G.M. and S. Schnell, 1994. Effect of increasing atmospheric methane concentration on ammonium inhibition of soil methane consumption. Nature, 370: 282-284.

Kinman, R.N., D.L. Nutini, J.J. Walsh, W.G. Vogt, J. Stamm and J. Rickabaugh, 1987. Gas enhancement techniques in landfill simulators. Waste Manage. Res., 5: 13-25.

Kravchenko, I., P. Boeckx, V. Galchenko and O. van Cleemput, 2002. Short- and medium-term effects of NH_{4+} on CH_4 and N_2O fluxes in arable soils with a different texture. Soil Biol. Biochem., 34: 669-678.

Kriofske, K., 1998. The use of polyethylene geomembranes in lining and gas collection covers for manure lagoons. Manure Management Conference, February 10-12, 1998, Iowa State University, Ames, Iowa, http://infohouse.p2ric.org/ref/21/20536.htm.

Kumaraswamy, S., B. Ramakrishnan and N. Sethunathan, 2001. Methane production and oxidation in an anoxic rice soil as influenced by inorganic redox species. J. Environ. Qual., 30: 2195-2201.

LaDeau, S.L. and J.S. Clark, 2001. Rising CO_2 levels and the fecundity of forest trees. Science, 292: 95-98.

Le Mer, J. and P. Roger, 2001. Production, oxidation, emission and consumption of methane by soils: A review. Eur. J. Soil Biol., 37: 25-50.

Leach, A., 1990. Landfill gas abstraction. Proceedings of international conference on landfill gas: energy and environment 90, October 21-44, 1990, Bournemouth, UK., pp: 204-222.

Lee, T.D., M.G. Tjoelker, D.S. Ellsworth and P.B. Reich, 2001. Leaf gas exchange responses of 13 Prairie grassland species to elevated CO_2 and increased nitrogen supply. New Phytol., 150: 405-418.

Leita, L., M. De Nobili, G. Muhlbachova, C. Mondini, L. Marchiol and G. Zerbi, 1995. Bioavailability and effects of heavy metals on soil microbial biomass survival during laboratory incubation. Biol. Fertil. Soils, 19: 103-108.

Lemna Technologies Inc., 2015. Gas collection covers. Lemna Technologies Inc., Vadnais Heights, MN., USA. http://www.lemnatechnologies.com/Lemna/gas-collection-covers.html.

Lessard, R., P. Rochette, E. Topp, E. Pattey, R.L. Desjardins and G. Beaumont, 1994. Methane and carbon dioxide fluxes from poorly drained adjacent cultivated and forest sites. Can. J. Soil Sci., 74: 139-146.

Liamsanguan, C. and S.H. Gheewala, 2008. LCA: A decision support tool for environmental assessment of MSW management systems. J. Environ. Manage., 87: 132-138.

Lindhout, P. and G. Pet, 1990. Effects of CO_2 enrichment on young plant growth of 96 genotypes of tomato (*Lycopersicon esculentum*). Euphytica, 51: 191-196.

Luxmoore, R.J., R.J. Norby and E.G. O'Neill, 1986. Seedling tree responses to nutrient stress under atmospheric CO_2 enrichment. Proceedings of the 18th IUFRO world congress forest plants and forest protection, (IUFROWCFPFP'86), International Union of Forestry Research, pp: 178-183.

Machadoa, S.L., M.F. Carvalho, J.P. Gourc, O.M. Vilar and J.C.F. do Nascimento, 2009. Methane generation in tropical landfills: Simplified methods and field results. Waste Manage., 29: 153-161.

Matthews, E., 1994. Nitrogenous fertilizers: Global distribution of consumption and associated emissions of nitrous oxide and ammonia. Global Biogeochem. Cycles, 8: 411-439.

Maurice, C., M. Ettala and A. Lagerkvist, 1999. Effects of leachate irrigation on landfill vegetation and subsequent methane emissions. Water Air Soil Pollut., 113: 203-216.

McBean, E., F. Rovers and G. Farquhar, 1995. Solid Waste Landfill Engineering and Design. Prentice Hall PTR, NJ, USA.

McGrath, R.J. and I.G. Mason, 2004. An observational method for the assessment of biogas production from an anaerobic waste stabilisation pond treating farm dairy wastewater. Biosyst. Eng., 87: 471-478.

Mishra, S.R., K. Bharati, N. Sethunathan and T.K. Adhya, 1999. Effects of heavy metals on methane production in tropical rice soils. Ecotoxicol. Environ. Saf., 44: 129-136.

Mohanty, S.R., K. Bharati, N. Deepa, V.R. Rao and T.K. Adhya, 2000. Influence of heavy metals on methane oxidation in tropical rice soils. Ecotoxicol. Environ. Saf., 47: 277-284.

Mora-Naranjo, N., J.A. Meima, A. Haarstrick and D.C. Hempel, 2004. Modelling and experimental investigation of environmental influences on the acetate and methane formation in solid waste. Waste Manage., 24: 763-773.

Moreira, J.M.L., E.R. da Silva and G. Candiani, 2015. Approximate estimation of landfill emissions considering methane oxidation. Open Waste Manage. J., 8: 12-20.

Mortensen, L.M., 1987. Review: CO_2 enrichment in greenhouses. Crop responses. Sci. Hortic., 33: 1-25.

Murrell, J.C., I.R. McDonald and B. Gilbert, 2000. Regulation of expression of methane monooxygenases by copper ions. Trends Microbiol., 8: 221-225.

Nguyen, H.H.T., A.K. Shiemke, S.J. Jacobs, B.J. Hales, M.E. Lidstrom and S.I. Chan, 1994. The nature of the copper ions in the membranes containing the particulate methane Monooxygenase from *Methylococcus capsulatus* (bath). J. Biol. Chem., 269: 14995-15005.

Nielsen, A.K., K. Gerdes and J.C. Murrell, 1997. Copper-dependent reciprocal transcriptional regulation of methane monooxygenase genes in *Methylococcus capsulatus* and *Methylosinus trichosporium*. Mol. Microbiol., 25: 399-409.

Niklaus, PA., M. Wohlfender, R. Siegwolf and C. Korner, 2001. Effects of six years atmospheric CO_2 enrichment on plant, soil and soil microbial C of a calcareous grassland. Plant Soil, 233: 189-202.

Oren, R., D.S. Ellsworth, K.H. Johnsen, N. Phillips and B.E. Ewers *et al*., 2001. Soil fertility limits carbon sequestration by forest ecosystems in a CO_2-enriched atmosphere. Nature, 411: 469-472.

Paradise, C.J. and R.J. Cyr, 1995. Carbon Dioxide Uptake in Plants: A Computer-Aided Experimental System. In: Tested Studies for Laboratory Teaching, Goldman, C.A. (Ed.)., Vol. 16, Association for Biology Laboratory Education (ABLE), Springfield, MO., pp: 99-114.

Park, J.W. and H.C. Shin, 2001. Surface emission of landfill gas from solid waste landfill. Atmos. Environ., 35: 3445-3451.

Permavoid Ltd., 2015. Virtual curtain gas migration system. http://www.permavoid.co.uk/sectors/gas_migration/introduction.html

Peterjohn, W.T., J.M. Melillo, P.A. Steudler, K.M. Newkirk, F.P. Bowles and J.D. Aber, 1994. Responses of trace gas fluxes and n availability to experimentally elevated soil temperatures. Ecolog. Applic., 4: 617-625.

Phelps, P.A., S.K. Agarwal, G.E. Speitel Jr. and G. Georgiou, 1992. *Methylosinus trichosporium* OB3b mutants having constitutive expression of soluble methane monooxygenase in the presence of high levels of copper. Applied Environ. Microbiol., 58: 3701-3708.

Prather, M., R. Derwent, D. Ehalt, P. Fraser, E. Sanhueza and X. Zhau, 1996. Other Trace Gases and Atmospheric Chemistry. In: Climate Change 1995: The Science of Climate Change, Houghton, J.T., L.G.M. Filho, B.A. Callandar, N. Harriss, A. Kattenberg and K. Maskell (Eds.)., Cambridge University Press, Cambridge, Uk, pp: 77-126.

Prieme, A. and S. Christensen, 1997. Seasonal and spatial variation of methane oxidation in a Danish spruce forest. Soil Biol. Biochem., 29: 1165-1172.

Rada, E.C., I.A. Istrate and M. Ragazzi, 2009. Trends in the management of residual municipal solid waste. Environ. Technol., 30: 651-661.

Rada, E.C., M. Ragazzi and V. Torretta, 2013. Laboratory-scale anaerobic sequencing batch reactor for treatment of stillage from fruit distillation. Water Sci. Technol., 67: 1068-1075.

Radoglou, K.M., P. Aphalo and P.G. Jarvis, 1992. Response of photosynthesis, stomatal conductance and water use efficiency to elevated CO_2 and nutrient supply in acclimated seedlings of *Phaseolus vulgaris* L. Ann. Bot., 70: 257-264.

Reinhart, D.R., 1996. Full-scale experiences with leachate recirculating landfills: Case studies. Waste Manage. Res., 14: 347-365.

Reinhart, D.R. and A.B. Al-Yousfi, 1996. The impact of leachate recirculation on municipal solid waste landfill operating characteristics. Waste Manage. Res., 14: 337-346.

Sasek, T.W., E.H. DeLucia and B.R. Strain, 1985. Reversibility of photosynthetic inhibition in cotton after long-term exposure to elevated CO_2 concentrations. Plant Physiol., 78: 619-622.

Scheutz, C. and P. Kjeldsen, 2004. Environmental factors influencing attenuation of methane and hydrochlorofluorocarbons in landfill cover soils. J. Environ. Qual., 33: 72-79.

Scheutz, C., J. Bogner, J.P. Chanton, D. Blake, M. Morcet, C. Aran and P. Kjeldsen, 2008. Atmospheric emissions and attenuation of non-methane organic compounds in cover soils at a French landfill. Waste Manage., 28: 1892-1908.

Scheutz, C., P. Kjeldsen, J.E. Bogner, A. De Visscher and J. Gebert *et al*., 2009. Microbial methane oxidation processes and technologies for mitigation of landfill gas emissions. Waste Manage. Res., 27: 409-455.

Scotford, I.M. and A.G. Williams, 2001. Practicalities, costs and effectiveness of a floating plastic cover to reduce ammonia emissions from a pig slurry lagoon. J. Agric. Eng. Res., 80: 273-281.

Sellers, P.J., F.G. Hall, H. Margolis, B. Kelly and D. Baldocchi *et al*., 1995. The boreal ecosystem-atmosphere study (BOREAS): An overview and early results from the 1994 field year. Bull. Am. Meteorol. Soc., 76: 1549-1577.

Sionit, N., B.R. Strain and H.A. Beckford, 1981. Environmental controls on the growth and yield of Okra. I. Effects of temperature and CO_2 enrichment at cool temperatures. Crop Sci., 21: 855-888.

Sitaula, B.K., L.R. Bakken and G. Abrahamsen, 1995. CH_4 uptake by temperate forest soil: Effect of N input and soil acidification. Soil Biol. Biochem., 27: 871-880.

Smith, D.S., T.E. Huxman, S.F. Zitzer, T.N. Charlet and D.C. Housman *et al*., 2000. Elevated CO_2 increases productivity and invasive species success in an arid ecosystem. Nature, 408: 79-82.

Sommerfeld, R.A., A.R. Mosier and R.C. Musselman, 1993. CO_2, CH_4 and N_2O flux through a Wyoming snowpack and implications for global budgets. Nature, 361: 140-142.

Spokas, K.A. and J.E. Bogner, 2011. Limits and dynamics of methane oxidation in landfill cover soils. Waste Manage., 31: 823-832.

Stein, V.B. and J.P.A. Hettiaratchi, 2001. Methane oxidation in three alberta soils: Influence of soil parameters and methane flux rates. Environ. Technol., 22: 101-111.

Steinkamp, R., K. Butterbach-Bahl and H. Papen, 2001. Methane oxidation by soils of an N limited and N fertilized spruce forest in the black forest, Germany. Soil Biol. Biochem., 33: 145-153.

Steudler, P.A., R.D. Bowden, J.M. Melillo and J.D. Aber, 1989. Influence of nitrogen fertilization on methane uptake in temperate forest soils. Nature, 341: 314-316.

Strain, B.R. and F.A. Bazzaz, 1983. Terrestrial Plant Communities. In: CO_2 and Plants: The Response of Plants to Rising Levels of Atmospheric Carbon Dioxide, Lemon, E.R. (Ed.). Westview Press for the American Association for the Advancement of Science, Boulder, CO., pp: 177-222.

Sulisti, I.A. Watson-Craik and E. Senior, 1996. Studies on the co-disposal of *o*-cresol with municipal refuse. J. Chem. Technol. Biotechnol., 65: 72-80.

Sundh, I., C. Mikkela, M. Nilsson and B.H. Svensson, 1995a. Potential aerobic methane oxidation in a Sphagnum-dominated peatland-controlling factors and relation to methane emission. Soil Biol. Biochem., 27: 829-837.

Sundh, I., P. Borga, M. Nilsson and B.H. Svensson, 1995b. Estimation of cell numbers of methanotrophic bacteria in boreal peatlands based on analysis of specific phospholipid fatty acids. FEMS Microbiol. Ecol., 18: 103-112.

Takeguchi, M. and I. Okura, 2000. Role of iron and copper in particulate methane monooxygenase of *Methylosinus trichosporium* OB3b. Catal. Surv. Jpn., 4: 51-63.

Thomas, J.F., C.D. Raper, C.E. Anderson and R.J. Downs, 1975. Growth of young tobacco plants as affected by carbon dioxide and nutrient variables. Agron. J., 67: 685-689.

Tlustos, P., T.W. Willison, J.C. Baker, D.V. Murphy, D. Pavlikova, K.W.T. Goulding and D.S. Powlson, 1998. Short-term effects of nitrogen on methane oxidation in soils. Biol. Fertil. Soils, 28: 64-70.

Topp, E. and R.S. Hanson, 1991. Methane Oxidizing Bacteria. In: Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides and Halomethanes, Rogers, J.E. and W.B. Whitman (Eds.). American Society for Microbiology, Washington DC., ISBN-13: 9781555810351.

Topp, E. and E. Pattey, 1997. Soils as sources and sinks for atmospheric methane. Can. J. Soil Sci., 77: 167-177.

Tremblay, N., S. Yelle and A. Gosselin, 1987. Effects of CO_2 enrichment, nitrogen and phosphorus fertilization on growth and yield of celery transplants. HortScience, 22: 875-876.

Tremblay, N. and A. Gosselin, 1998. Effect of carbon dioxide enrichment and light. HortTechnology, 8: 524-528.

Trulli, E., V. Torretta, M. Raboni and S. Masi, 2013. Incineration of pre-treated Municipal Solid Waste (MSW) for energy co-generation in a non-densely populated area. Sustainability, 5: 5333-5346.

Tuskan, G.A. and M.E. Walsh, 2001. Short-rotation woody crop systems, atmospheric carbon dioxide and carbon management: A U.S. case study. For. Chronicle, 77: 259-264.

Uprety, D.C., R.S. Mishra and Y.P. Abrol, 1995. Effect of elevated CO_2 on the photosynthesis, growth and water relation of *Brassica* species under moisture stress. J. Agron. Crop Sci., 175: 231-237.

Uprety, D.C., N. Dwivedi and R. Mohan, 1998. Characterization of CO_2 responsiveness in a *Brassica oxycamp* interspecific hybrid. J. Agron. Crop Sci., 180: 7-13.

Uprety, D.C., S. Kumari, N. Dwivedi and R. Mohan, 2000a. Effect of elevated CO_2 on the growth and yield of rice. Indian J. Plant Physiol., 5: 105-107.

Uprety, D.C., S.C. Garg, M.K. Tiwari and A.P. Mitra, 2000b. Crop responses to elevated CO_2: Technology and research (Indian studies). Global Environ. Res., 3: 155-167.

Uprety, D.C., N. Dwivedi, V. Jain and R. Mohan, 2002. Effect of elevated carbon dioxide concentration on the stomatal parameters of rice cultivars. Photosynthetica, 40: 315-319.

Urban, L., 1994. Effect of high-pressure mist and daytime continuous CO_2 enrichment on leaf diffusive conductance, CO_2 fixation and production of rosa hybrida plants grown on rockwool. Acta Horticulturae, 361: 317-324.

Vaccari, M., V. Torretta and C. Collivignarelli, 2012. Effect of improving environmental sustainability in developing countries by upgrading solid waste management techniques: A case study. Sustainability, 4: 2852-2861.

Van Der Nat, F., J. de Brouwer, J. Middelburg and H.J. Laanbroek, 1997. Spatial distribution and inhibition by ammonium of methane oxidation in intertidal freshwater marshes. Applied Environ. Microbiol., 63: 4734-4740.

Wand, S.J.S., G.Y.F. Midgley, M.H. Jones and P.S. Curtis, 1999. Responses of wild C_4 and C_3 grass (poaceae) species to elevated atmospheric CO_2 concentration: A meta-analytic test of current theories and perceptions. Global Change Biol., 5: 723-741.

Warith, M.A., W. Zekry and N. Gawri, 1999. Effect of leachate recirculation on municipal solid waste biodegradation. Water Qual. Res. J. Can., 34: 267-280.

Westlake, K., 1995. Landfill. In: Waste Treatment and Disposal, Hester, R.E. and R.M. Harrison (Eds.). The Royal Society of Chemistry, Herts, Cambridge.

Whalen, S.C., W.S. Reeburgh and K.A. Sandbeck, 1990. Rapid methane oxidation in a landfill cover soil. Applied Environ. Microbiol., 56: 3405-3411.

Wheless, E. and G. Wiltsee, 2001. Demonstration test of the capstone microturbine on landfill gas. Proceedings of the 24th Annual Landfill Gas Symposium, March 19-22, 2001, Dallas, Texas.

Williams, P.T., 2005. Waste Treatment and Disposal. 2nd Edn., John Wiley and Sons, New York, USA., pp: 171-244.

Wilson, S.A. and A. Shuttleworth, 2002. Design and performance of a passive dilution gas migration barrier. http://www.nce.co.uk/design-and-performance-of-a-passive-dilution-gas-migration-barrier/712380.article.

Wong, S.C., 1979. Elevated atmospheric partial pressure of CO_2 and plant growth. I. Interactions of nitrogen nutrition and photosynthetic capacity in C_3 and C_4 plants. Oecologia, 44: 68-74.

Woodrow, L., B. Grodzinski and A. Liptay, 1987. The effects of CO_2 enrichment and ethephon application on the production of tomato transplants. Acta Horticulturae, 201: 133-140.

Xin, J.Y., J.R. Cui, J.Z. Niu, S.F. Hua, C.G. Xia, S.B. Li and L.M. Zhu, 2004. Production of methanol from methane by methanotrophic bacteria. Biocatal. Biotransform., 22: 225-229.

Zak, D.R., K.S. Pregitzer, P.S. Curtis, J.A. Teen, R. Fogel and D.L. Randlett, 1993. Elevated atmospheric CO_2 and feedback between carbon and nitrogen cycles. Plant Soil, 151: 105-117.

Zhang, C., H. Su, J. Baeyens and T. Tan, 2014. Reviewing the anaerobic digestion of food waste for biogas production. Renew. Sustain. Energy Rev., 38: 383-392.

Zhang, H., P. He and L. Shao, 2008. Methane emissions from MSW landfill with sandy soil covers under leachate recirculation and subsurface irrigation. Atmos. Environ., 42: 5579-5588.