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Quantification of Methane Emission from Dhaka City Landfill Solid Wastes and Its Potentiality

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ABSTRACT

Solid *waste management currently considered as one of the most alarming problem of Dhaka city. Huge amount of wastes emits significant amount of green house gas, especially methane gas. Green house methane gas is contributing to global warming which is the principle cause of climate change. AMS-III.G CDM method was selected in this paper to calculate the baseline methane emission from solid wastes. Using this 1st order decay model of estimation, it was found that the methane emission from Dhaka city solid waste was about 319027 tons in a year. When certain amount of wastes was left to decay, it was found that emission decreases exponentially to 51755 tons after ten years. It was also found that when waste dumping continues for years, emission will increase exponentially to 1473680 tons after ten years. It was recommended to implement a Clean Development Mechanism (CDM) through an emission reduction solid waste management project and recover this amount of green house gas from escaping into environment. Efficient waste management, global warming lessening and sustainable development will take place through this mechanism.*

INTRODUCTION

It is well recognized that Bangladesh is one of the most adversely affected countries to climate change. Low economic strength, inadequate infrastructure, low level of social development, lack of institutional capacity and a higher dependency on the natural resource base make the country more vulnerable to climate impetus in different sectors like agriculture, biodiversity, fishery, forest, energy etc. Strategies to cope climate change have become vital for sustainable development in different sectors. The Clean Development Mechanism (CDM) of the Kyoto Protocol operates on the premise that the atmosphere is a well-mixed global system, and a reduction in greenhouse gas (GHG) emissions in one country benefits all countries. It also outlines the importance of investment in sustainable environmentally beneficial projects in the developing world. Through the CDM, industrialized countries with GHG reduction obligations under the Protocol can provide financial support for GHG reduction projects in developing countries. For instance, Landfill gas (LFG) CDM projects offer the chance to reduce GHG emissions while improving landfill management practices using revenue generated by the sale of reductions. Therefore, establishing an inventory of GHG emissions-specially of methane emissions in different sectors of a developing country like Bangladesh has become very essential.

Dhaka is a rapidly growing mega city. It has been facing several severe challenges with its booming population of around 15.0 million that is nearly 10% of countries total population. The city, its municipalities and its adjoining urban areas have a waste generation rate is 0.485 kg/person/day (Alamgir et al. 2007). With the annual population growth rate of 1.8 % (BBS, 2011), the management of solid waste has become one of the most alarming and immediate problem of Dhaka city. Dhaka city produces wastes more than 4000 tons/day of which around 50-60% finally reaches to landfill (waste concerns, 2009) after losses due to roadside, gutter/drain or illegal dumping. The crude open dumping of solid wastes is in no way a sanitary land filling and thus is not scientific and hygienic. The entire unmanaged wastes are creating many environmental nuisance, health hazards and economic losses.

On the other hand, Landfill gas (LFG) can be an energy source. Landfill gas is typically comprised of approximately 50 percent methane and 50 percent carbon dioxide and trace amount of other gases. Landfill Methane gas which is a potential source of greenhouse gas emission can be flared to be converted to other less harmful gases or captured and used in energy purposes which may significantly reduce emission. Thus the study of methane gas emission has given importance globally.

However, Lack of availability of data concerning methane generation from solid waste of Dhaka and selecting appropriate methodology is a big problem. Nevertheless, some previous studies attempted to quantify methane emission from Dhaka city solid waste.
Rahman et. al.(2010) found methane emission from Dhaka city solid waste using Inter Governmental panel of Climate Change (IPCC) guideline tier-1 to be 26.89 Giga gram or 26.89 Million tons. Another study by Mostain (2009) found methane emission from Dhaka city corporation waste is 118.82 Gigagrams /year.

However, in all previous studies, real methane emission inventories that can be potentially recovered or used in a CDM project were absent. No such study accounted for the waste existed in landfill for projecting future emissions.

METHODOLOGY

Intergovernmental Panel on Climate Change (IPCC) and United Nations Framework Convention on Climate Change (UNFCCC) recommend a number of methodologies to estimate the amount of methane gas produced in the landfill. Application of these methodologies depends on the existing types of landfills and characteristics of the project. From our countries perspectives of uncontrolled open dumping, we have selected AMS-III.G; version 01 (dated 03 March 2006) method to quantify the baseline methane emission. This is an approved methodology by the UNFCCC CDM Executive board.
As per the methodology “The baseline scenario is the situation where, in the absence of the project activity, biomass and other organic matter are left to decay within the project boundary and methane is emitted to the atmosphere. The baseline emissions are the amount of methane from the decay of the biomass content of the waste treated in the project activity. The Yearly Methane Generation Potential calculated using the first order decay model based on the discrete time estimate method of the IPCC Guidelines, as described in category AMS III-G. Baseline emissions shall exclude methane emissions that would have to be removed or combusted to comply with national or local safety requirement or legal regulations.”
Yearly Methane Generation Potential:

$$MBy = \frac{16}{12} \times F \times DOC_f \times MCF \times \sum_{x=1}^{y} \sum_{j=A}^{D} A_{j,x} \times DOCj \times \left(1 - e^{-k_j}\right) \times e^{-k_j\,(y-x)} \tag{1}$$

where,
MB_y is the amount of methane generated during years y
F is fraction of methane in the landfill gas
DOC_j is percent of degradable organic carbon (by weight) in the waste type j
DOC_f is fraction of DOC dissimilated to landfill gas
MCF is Methane Correction Factor
$A_{j,x}$ is amount of organic waste type j landfilled in the year x (/year)
k_j is decay rate for the waste stream type j
x is year since the landfill started receiving wastes: x runs from the first year of landfill operation (x=1) to the year for which emissions are calculated (x=y)
y is year for which LFG emissions are calculated

In our calculation for DCC waste, the value of some parameters was taken from secondary sources, while others have the standard values. Solid waste disposal sites (SWDS) produce GHG’s where 40 – 50% is methane gas. So fraction of methane in landfill gas, F was taken 0.5 (IPCC default). Fraction of degradable organic carbon (DOC) that can decompose (DOC_f) is very high in Bangladesh. For Dhaka city SWDS, the value was taken 0.76 (Waste Concerns, 2009).The methane correction factor (MCF) accounts for the fact that unmanaged SWDS produce less methane from a given amount of waste than managed SWDS because a larger fraction of waste decomposes aerobically in the top layers of unmanaged SWDS. Present Dhaka city SWDS are unmanaged and shallow (depth is less than 4.0 meter),so the value of MCF was taken 0.4 (IPCC).

Households, construction rubbish, market areas, street sweeping, garden trimming, industrial units, commercial offices, restaurants and hospitals are the main sources of wastes in Dhaka City. It was difficult to measure the current amount of waste generation because per capita waste generation fluctuates for different classes of people of Dhaka city and for floating people as well. Moreover, dry and wet season waste production varies also. Therefore, different literature has different amount of waste which is shown in Table 1. Not all waste that produces finally reach to landfills as waste collection efficiency by DCC authority is not at satisfactory level due to lack of ability and proper management. Table 1 also shows waste collection efficiency. The remaining wastes causing environment nuisance such as scattered garbage, offensive odor, drain clogging, water pollution and mosquitoes.

Table 1 Amount of waste Land filled

Source	Total Waste Generation (tons/day)	Waste Collection Efficiency	Average Waste Generation Taken (tons/day)	Average Waste Collection Efficiency Taken	Amount of Waste Land filled (tons/year)
DCC[1]	4000	60%	3500	55%	702625
Matuail Report[2]	3200	50%			
Waste Concerns[3]	3768	60%			

[1] Dhaka City Corporation (2001)
[2] Matuail Report, BRTC, BUET (2008)
[3] Waste Database (2009).

The major type of wastes (j) that are responsible for methane emission are food waste, paper and textile waste, park and garden waste and wood waste. Fraction of waste components in total wastes of Dhaka city with their respected values of percent of degradable organic carbon (DOC_j) and decay rate (k_j) are shown on Table 2. Note that decay rate k_j considered here (as perspective from our country) from IPCC are for the regions with Mean Annual Temperature (MAT) > 20° C and Mean Annual Precipitation (MAP) > 1000 mm, and wastes considered to be wet. Decay rates are - low for dry wastes, and less for low temperatures (IPCC 2006). Methane emissions in one year after waste dumping were calculated and shown in the Table 2.

Table 2 Fraction of waste categories and methane emission

Waste component	Fraction of total waste[1] (%)	Amount of waste, A_j (tons/year)	DOCj[2] (%)	Decay rate[2] (k_j)	Baseline Methane emission in one year (tons)	Total Baseline methane emission in one year (tons)
Food and vegetable waste	68.5	481298.13	15	0.231	301791.1	310279
Paper and textiles	11.6	81504.5	40	0.023	15023.454	
Garden and park waste	3.74	26278.18	17	0.023	2058.60	
Wood waste	0.16	1124.2	30	0.023	155.415	

1. Adapted from Ahmed M. F.et al (2000) and Dhaka city corporation
2. IPCC 2006 Guidelines for National Greenhouse Gas Inventories (adapted from Volume 5).

Similarly when a certain amount of waste was left to decay, baseline methane emissions were calculated for next ten years for different types of wastes and total wastes as well.

RESULT ANALYSIS AND DISCUSSION

Using the 1st order decay model, we have found methane emission 310279 tons or 310.28 Gigagrams in a year (or every year when waste generation or population is constant). It was found that when no more waste was added after one and subsequent years, emissions decreased with time and after 10 years emission was found 51755 tons (Figure 1).The reason may be that during early period, organic content of the wastes is high. With time, wastes decompose gradually and emission decreases subsequently.

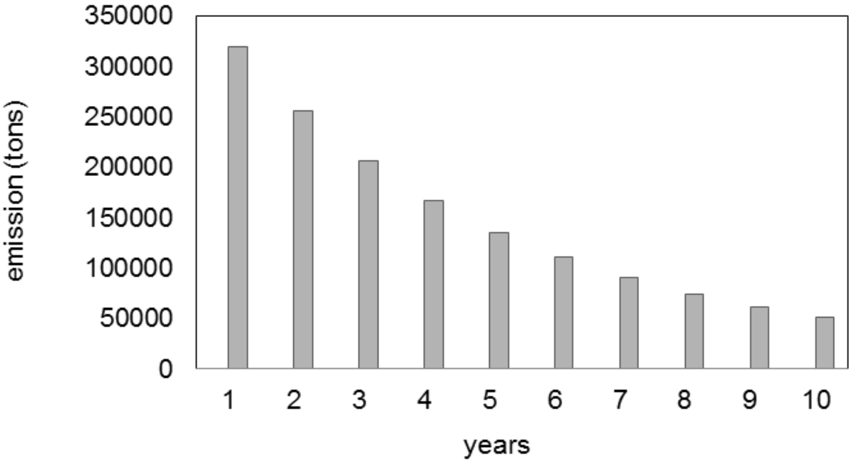

Figure 1 Methane emission from waste

On the other hand, baseline methane emission rises with time when the waste dumping continues for years. That is, when a landfill receives wastes for years, emission potential of methane increases and after 10 years total baseline methane emission was found 1473680 tons (Figure 2).

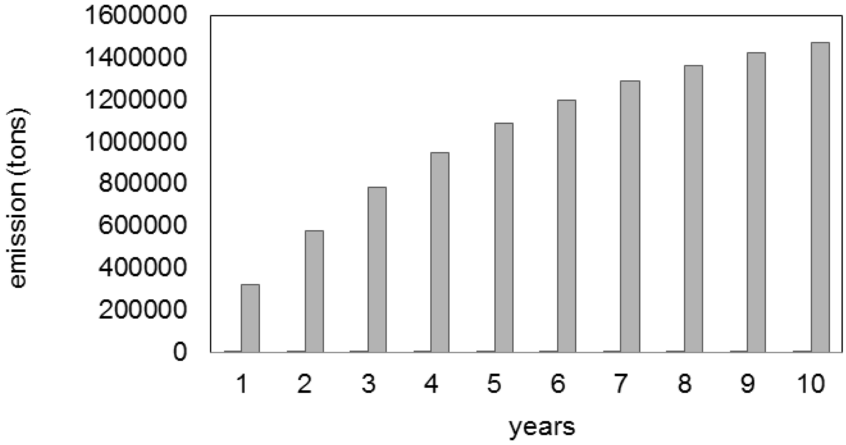

Figure 2 Cumulative methane emissions from waste

In all emission calculations in this paper, waste generation and population assumed to be constant in predicted different years due to lack of availability of data. Therefore, it is recommended to work on future waste generation and population fluctuation to account for better inventories in for future years.

Global Warming and Potential Methane Recovery Projects

The estimated methane emission from landfill solid waste will escape into environment and contribute to global warming. This huge amount of methane gas can be potentially captured and used in a sustainable project. With the global warming potency of methane which is 21, it was calculated (1473680 tons of methane/year X 21 eCO2/ton of Methane) that around 31 Million tons of eCO_2 (equivalent carbon dioxide) may produce from Dhaka city landfill solid waste in the next 10 years. Therefore, to attain their emission reduction targets -developed countries can invest in a waste management CDM project in Dhaka city that will recover methane gas. According to present global carbon market we can attract about 620 Million USD (31 eCO_2 × 20 \$/eCO2) foreign investments in the next 10 years. Through the investment, we can utilize methane gas and implement CDM sustainable development projects. Several sectors are available to use generated methane gas. Project choice depends on the available technology, fund, priority etc. Some example of projects-

- Generation of electricity. Electricity generation using a reciprocating engine, steam turbine, or gas turbine is a common end use of captured gas. In larger applications, high efficiency

Combined Cycle Gas Turbines (CCGT) can be operated successfully. Micro turbines can also be used for electricity generation from CH_4.

- Generation of hot water or steam from boilers (onsite and offsite). This project type involves the generation of hot water or steam using industrial or commercial boilers. Captured CH4 can be used at any type of boiler whether located onsite or offsite
- Delivering captured methane into a pipeline system or simple conversion to Compressed Natural Gas (CNG) of Liquefied Natural Gas (LNG)
- Other direct uses. Other direct uses consist of heating (e.g., furnaces, kilns, engines, space heaters) for various commercial and industrial uses, onsite leachate evaporation systems, and cooling (e.g., chillers, air conditioning), transportation fuel.

CONCLUSION

Climate change is a long-term problem that requires long-term solutions. Reducing emissions will decrease both the rate of change and the magnitude of those changes in climate and their related impacts. Clean development Mechanism offers an opportunity to reduce emission at lower costs by the host countries along with sustainable development. Therefore, it is imperative to generate GHG emission inventory for all different sectors like Agriculture, Forestry, Transport, Construction, Industry, Mining etc. Bangladesh has huge opportunity to achieve sustainable development at those different sectors. In this paper it was found from the inventory of methane gas that around 31 Million eCO_2 can be reduced from a methane recovery solid waste management project in Dhaka city. A spreadsheet analysis of the above methane emission model was developed that can be used to calculate methane emission and global warming potency for any year and any type of waste. Therefore, Government and private sectors have to take proper initiatives to upgrade the landfill management practice to accommodate ever increasing waste of Dhaka city that will enable both global warming lessening and sustainable development.

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Comparing LCA and CDM Methods – Two Ways to Calculate Greenhouse Gas Emission Reduction Due to Organic Waste Treatment

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ABSTRACT

The research study presented in this paper compares three treatment scenarios for organic municipal waste in the context of Cochabamba, Bolivia and quantifies greenhouse gas emissions using two methods. The options are: 1) disposal of organic waste at a landfill; 2) treatment by anaerobic digestion; 3) treatment by composting. Two different approaches to quantify greenhouse gas emissions of the waste treatment scenarios were applied: a) the methodology used in CDM projects to calculate Certified Emission Reductions (CERs); and b) the Life Cycle Analysis technique used in research and academia. Both methods show that treatment of organic waste has a large emission reduction potential when compared to landfill disposal. CDM methodology underestimates the effective emissions. Additionally, for composting the default value only seems to be accurate when good operational practices are guaranteed. For the case of Cochabamba the unfavourable climatic conditions result in a rather low potential to obtain certified emission reductions.

INTRODUCTION

In many developing countries, methane (CH_4) emissions from waste management and specifically from landfills is the largest anthropogenic source of atmospheric CH_4 (Spokas et al., 2006). These emissions occur when solid waste containing organic matter is disposed at a landfill. Under anaerobic conditions the biodegradable fraction undergoes microbial decomposition and forms landfill gas mainly containing CH_4 and CO_2 (Obersteiner et al., 2007). With increasing thickness of the waste layer at the dump site the potential for anaerobic decomposition increases and thereby also the potential for CH_4 generation. In uncontrolled dump sites, landfill gas is neither collected or flared and therefore released in an uncontrolled way into the atmosphere. Matthews and Themelis (2007) estimate that landfills contribute between 5-10% of global methane emissions or about 10% of the anthropogenic fraction. This indicates that with improved waste management and appropriate waste treatment technologies there is a large potential to contribute to a reduction of global greenhouse gas emissions.

In recent decades municipal solid waste management has become a major priority for municipalities and local governments and one of their major challenges (Zurbrügg, 2012). Especially in low- and mid-income countries where waste collection service coverage is low and appropriate treatment is lacking, this leads to water, land and air pollution, putting people’s health and the environment at risk (CWG, 2008). Current trends of increasing urbanization and economic development in many developing countries suggest that the problem will intensify as more waste is generated and with increasing coverage also more waste is transported and landfilled. Furthermore the trend towards controlled landfilling may also results in higher rates of CH_4 generation and emissions than the previous open-dumping and burning practice. Therefore a close look must be directed towards alternative treatment for the organic fraction of waste while focusing on a quantification of potential greenhouse gas emissions reduction as compared to landfilling. Comparison of different treatment options for organic waste and their respective CH_4 emissions can be a critical factor influencing decision making for the implementation of project activities in waste management.

In 1997 the Kyoto Protocol was developed with the goal to reduce greenhouse gas emissions until the year 2012. This legally binds countries that ratified the Protocol to reduce their greenhouse gas emissions by 5.2% relative to the year 1990 (UNFCCC, 2011). To support countries in limiting or reducing their greenhouse gas emissions and to encourage the private sector and developing countries to contribute, the Kyoto Protocol introduced the Clean Development Mechanism (CDM) (UNFCCC, 2010a). With the support of the Clean Development Mechanism developing countries can obtain finances to support emission reduction projects. through the sale of Certified Emission Reductions (CERs) These CERs, each equivalent to one tone of CO_2, can be traded and used by industrialized countries to meet a part of their emission reduction targets (UNFCCC, 2010a). To estimate CO_2 reduction potential, the United Nations Framework Convention on Climate Change (UNFCCC) provides approved methodologies and guidelines. However, these methodologies only consider greenhouse gas emissions and neglect any further environmental impact such as consumption of scarce resources or emission of pollutants others than greenhouse gases. On the other hand Life Cycle Assessment (LCA) offers the possibility to assess the entity of the environmental impact of a process or a product and therefore allows to take further criteria other than climate change into account.

The research study presented in this paper compares three different treatment scenarios for organic municipal waste in the context of Cochabamba, Bolivia and quantifies their greenhouse gas emissions using two different methodologies. The first is the conventionally used and approved methodologies used in CDM projects to calculate the expected Certified Emission Reductions (CERs). The other method is the Life Cycle Analysis technique, most frequently used in research and academia. The three waste treatment options compared are: 1) the disposal of organic waste at a landfill (which is considered the baseline); 2) the treatment by anaerobic digestion; 3) the treatment by composting. The two organic waste recycling options link to an ongoing technical cooperation project called *"Ecovecindarios"* led by the Swiss Foundation for Technical Cooperation (Swisscontact) in Cochabamba, Bolivia. As the project would like to benefit from financial support through the Clean Development Mechanism a detailed calculation of the possible greenhouse gas emission reductions for the two options is required.

Solid Waste Management in Cochabamba, Bolivia

Bolivia is a landlocked country in Central South America, with a surface area of 1’098’581 km^2 and a population of 10’118’683 inhabitants (CIA, 2011). The climate in Bolivia is dominated by the country’s largely variable altitude which ranges from 130 m.a.s.l. in the lowlands of the Amazon Basin up to 6’542 m.a.s.l.in the Andes mountain range. Despite its large natural resources, Bolivia is one of the poorest and least developed countries in Latin America. Cochabamba is the fourth largest city of Bolivia with a population of 625’429 inhabitants at an altitude of 2’558 m.a.s.l. with a sunny and moderate climate and with an average temperature of 17.6 °C. The climate is relatively dry, except during the rainy season between November and March where heavy rainfall events occur (INE, 2011). Collection and disposal of municipal waste in Cochabamba lays in the responsibility of the municipal services “Empresa Municipal de Servicios de Aseo (EMSA)”. Household waste is collected regularly on defined routes in the different neighborhoods of the city and then transported to the landfill K’ara K’ara south of the city. During the past years, waste disposal at the landfill has led to social conflicts given its insufficient management and the resulting health and environmental threats. Solutions are sought to reduce the amount of waste transported to K’ara K’ara for final disposal. Cochabamba has a waste production of 0.51 kg per inhabitant and day which amounts to about 319 tons of solid waste per day. The share of organic waste is 61% (66% considering the whole district) however, experiences of Swisscontact (2010) show that recovery of only 42% (134 t/day) of the organic fraction is feasible.

METHODS AND MATERIALS

This study compares a baseline of organic waste disposal at a landfill (scenario 0) with two organic waste treatment options, anaerobic digestion (AD) (scenario 1) and composting (scenario 2) Two methods are used to assess greenhouse gas emissions of the three scenarios. The first method is the one typically used in CDM projects which is approved by UNFCCC to calculate the expected Certified Emission Reductions (CERs). The second method is the Life Cycle Analysis technique.

UNFCCC and CDM Methods

The methodologies used from UNFCCC include the simplified methodologies for small scale projects which must meet the following criteria: for type I projects (Renewable Energy) the energy production potential of 15 MW shall not be exceeded and for type III projects (Waste Management) the emission reduction cannot go beyond the limit of 60’000 tons of CO_2 equivalent (CO_2e) per year. For project activities, where the emission reduction increases during the crediting period the project activities must remain under the limits of small scale projects every year during the crediting period (UNFCCC, 2011c). For the scenarios calculated in this study these conditions are met. The methods and tools used are as follows (more information on the methods is available at the UNFCCC web site).

- III.AO: Methane recovery through controlled anaerobic digestion
- III.F: Avoidance of methane emission through composting
- I.CT: Thermal energy production with or without electricity
- III.H: Methane recovery in wastewater treatment
- I.D: Grid connected renewable electricity generation
- Tool: methane emissions from disposal of waste at a solid waste disposal site
- Tool: project or leakage CO_2 emissions from fossil fuel consumption
- Tool: emission factor for an electricity system

The baseline scenario reflects all the emissions occurring without implementing any CDM project activity and is calculated with an equation (UNFCCC, 2010b) based on the First Order Decay Model (FOD). The results are in tons of CO_2 equivalents (tCO_2e) emitted. Projects such as composting or anaerobic digestion divert waste from disposal and therefor avoid this amount of CO_2e. Specific project emissions are then deducted to obtain the total emission reduction.

For composting and AD, project emissions comprise: a) emissions from incremental transport of waste (or co-substrate) in the year; b) emissions from incremental transport of compost (or digestate) in the year; c) emission from electricity or fossil fuel consumption for composting or AD in the year; d) methane emissions from composting or leakages from AD in the year; e) emission from treatment of leachate in the year.

If biogas from AD is used as a substitute for a fossil fuel energy source these additional emission reductions can be credited, whereby two options from combustion of biogas, generation of electricity or thermal energy generation, can be distinguished. In addition a calculation of the avoided emission of wastewater treatment was included as in the case of Bolivia co-substrate digestion from a slaughterhouse was envisaged.

Life Cycle Analysis Methods

In the Standard ISO 14040, the Life Cycle Assessment is defined as a methodology to assess the environmental aspects and significant environmental impacts related to a product or service. This service is defined in the "functional unit" and different processes providing the same service can be compared with regard to their environmental impact (Klöpffer and Grahl, 2009). The system boundary used is geographical site of the municipality of Cochabamba with its surrounding agricultural areas and the time frame is defined as 100 years, assuming that thereby in most of the long term emissions are included. Benefits to the environment by substituting goods that would have caused a certain environmental impact are also taken into account. These are: a) Renewable energy: as the produced biogas from anaerobic digestion can be used further as an energy source and substitutes energy from other sources; b) Organic fertilizer: as anaerobic digestion and composting produce an organic fertilizer that can be used on fields otherwise fertilized by inorganic fertilizers.

The overall environmental impact caused by emissions and resource consumption described in the inventory was assessed using the indicator IPCC 2007 GWP (for CO_2e emissions) and ReCiPe Mid/Endpoint method, version 1.05 (for environmental impact) (Forster et al., 2007; ReCiPe, 2011). This paper however only highlights the results concerning the CO2e emissions. A brief summary of the inventory for each option is described below. More details on inventories and ReCiPe results are described in Volkart (2011).

Inventory for Anaerobic Digestion: The infrastructure data is based on a fictional industrial biogas plant which includes a storage vessel, the main reactor and a gas storage reactor, a heat exchanger and a management building. The reactor is made out of steel and concrete and includes stirrers to ensure homogenization. Furthermore the system includes a heat exchanger to warm up the substrate before it is fed into the reactor. Additionally, a management and technology building is required, where offices, toilets, showers, as well as storages units are situated. The building is considered to be made of concrete, bricks and corrugated sheet metal (Hartmann, 2006). The inventory includes emissions and resource consumption from the construction and disposal of the plant. The operation of the biogas plant consumes energy mainly for pumping and stirring the substrate and to heat the substrate to operation temperature (Deublein and Steinhauser, 2011). The electric energy consumption is assumed to be covered by electricity from the national grid of Bolivia. In addition a calculation of the avoided emission of wastewater treatment was included as in the case of Bolivia co-substrate digestion from a slaughterhouse was envisaged. From the anaerobic digestion of organic waste, gaseous emission such as CH_4, N_2O and NH_3 occur and are considered in the inventory. Emissions occur mainly during the maturation process when fresh digestate is mixed with compost material and undergoes an aerobic process (Edelmann and Schleiss, 2001) but also during delivery, preparation and final solid matter separation of the substrate. Additionally methane may escape from leaks if the plant is not completely gastight (Gyalpo, 2010; UNFCCC, 2010c). This is assessed in a sensitivity analysis. The amount of produced biogas is calculated according to the composition of the organic waste in Cochabamba. For both uses, electricity and biogas, it is assumed that an equal energetic amount of either electricity from the grid or natural gas is substituted, providing credits for the avoided emissions from combustion of fossil fuel. With use of compost or digestate in agriculture, the effect of the introduced nutrients but also contaminants such as heavy metals is considered. To estimate the emissions from transportation, infrastructure of the trucks and the distance to the city market La Cancha (amount of combusted fuel) were calculated.

Inventory for Composting: The infrastructure consists of an open compost plant on concrete ground, including a leachate collection system. The area where the intensive composting process takes place under a roof and the maturation is considered to be under open sky. For the infrastructure of the compost plant all the emissions from the construction and disposal of the plant are taken into account (Edelmann and Schleiss, 2001). Energy is consumed from shredding and loading the waste, turning the windrows and maintaining the system for forced aeration. The electricity consumption is covered by electricity from the national grid while emissions from the fuel consumption are calculated considering the resource consumption as well as emissions from combustion. During composting some N_2O, NH_4, and CH_4 are emitted (Edelmann and Schleiss, 2001; Stucki, 2007) and are considered as emissions. The emissions from application of compost as well as emissions from transportation are assumed to be equal to the ones from the scenario of anaerobic digestion.

Inventory for Landfill: The inventory accounts for all emissions from the construction of a sanitary landfill including resource and energy consumption and emissions. The landfill is designed for the storage of 1.8 million m3 of waste and is assumed to be partly submerged below the existing surface and to rise above it after closure. The plant includes a leachate collection system as well as roads accessing the landfill (Doka, 2009). The waste disposed at the landfill causes gaseous emission, mainly consisting of CO_2 and CH_4. Biogenic CO_2 emissions are considered carbon-neutral however CH_4 emission and additional airborne pollutants are taken into consideration. Additionally, it is also considered that a fraction of the produced CH_4 is oxidized in the clay landfill cover and another part is flared. This is subtracted from the produced emissions. During operation of the landfill fuel is consumed by activities such as waste compaction, regular covering and maintenance of the leachate recovery system. All these activities are required because of the organic waste and are therefore all accounted towards the organic fraction. For transport of waste to the landfill, the trucks and fuel consumption are taken into account.

Data Collection

The calculations were done for project scenarios, in other words for a project which has not yet been implemented. Most site specific data was obtained from secondary sources relating to the current waste management and treatment experiences in Cochabamba. To complete the inventory for the three scenarios, different approaches for data collection were applied: Literature research was used to obtain information on the necessary material and energy flows of the waste treatment processes. Furthermore, different studies to assess the specific emissions of organic waste treatment were reviewed. During a field visit in Cochabamba, semi-structured interviews with stakeholders were conducted to obtain information about the current waste management as well as to assess the enabling environment for the implementation of the new project. Where applicable, data gaps were filled using default values suggested in the IPCC Guidelines for National Greenhouse Gas Inventories. Where available, processes from the Ecoinvent Database (Frischknecht and Rebitzer, 2005) were used either directly or then in a slightly modified manner.

RESULTS and DISCUSSION

LCA Results

Scenario 0: Considering the Global Warming Potential (GWP) assessed with the method IPCC 2007 GWP 100y the overall impact of landfill disposal is estimated to be a total of 868 kgCO$_2$e per ton of organic waste landfilled. The contribution of CH$_4$ emissions account for 97% of the total GWP, transportation for 1.87% (16.2 kgCO$_2$e) and infrastructure for 0.79% (6.87 kgCO$_2$e) (Fig. 1).

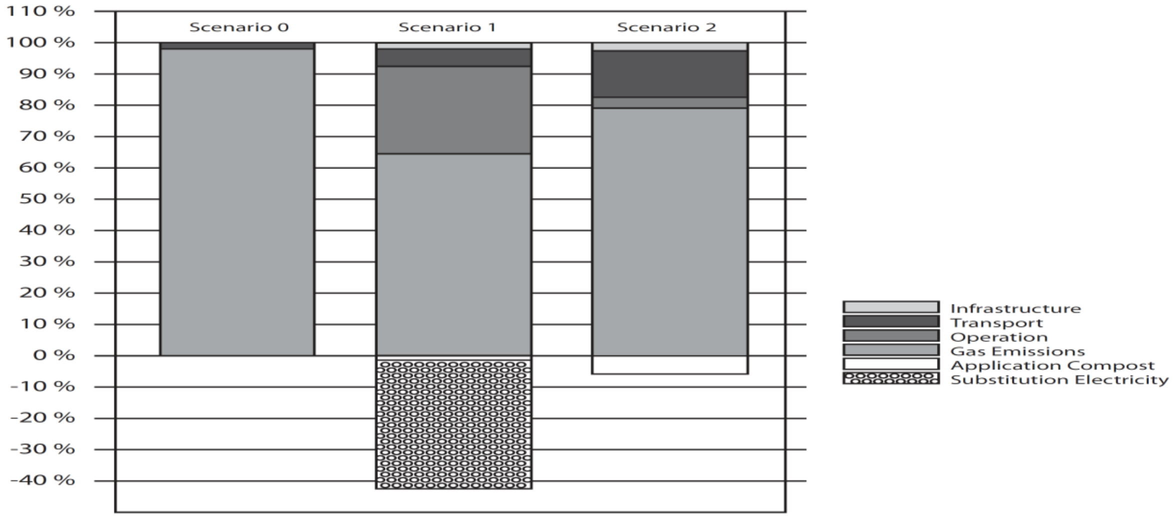

Figure 1: Percent of environmental impact by categories for the three scenarios (GWP IPCC 100y) (adapted after Volkart, 2011).

Scenario 1: The total emission from anaerobic digestion project activities (based on GWP) amount to 299.54 kgCO$_2$e per ton of organic waste treated. The largest share of emissions are caused by the operation of the plant (74.1 kgCO$_2$e) and the gaseous emissions of organic matter before, during (leakage), and after the biogas plant (192 kgCO$_2$e) (Fig.1). Except for the digestion of organic matter, in which case the total GWP is entirely caused by CH4 (151 kgCO$_2$e) and N2O (41 kgCO$_2$e), all other processes at the AD plant contribute almost exclusively by the emission of fossil fuel combustion. On the benefit side, the conversion of the gas to electricity and its use as a substitute to Bolivian electricity grid helps to avoid 125 kgCO$_2$e, while the application of compost causes a reduction of the overall emission by 6.54 kgCO$_2$e, mainly by avoided fossil fuel combustion. The net total emissions are therefore 168 kgCO$_2$e per ton of organic waste treated.

Scenario 2: For the compost scenario, total greenhouse gas emissions (GWP) account for 108 kgCO$_2$e. The largest impact is caused by gaseous emissions during the composting process with a contribution of 91 kgCO$_2$e. Of the 91 kgCO$_2$e, CH4 accounts for the largest share with 59.7 kgCO$_2$e while N2O contributes with 31.1 kgCO2e. Operation of the plant (4.18 kgCO$_2$e) and the infrastructure of the biogas plant (2.6 kgCO$_2$e) contribute only little to the total emissions (Fig.1). The emissions from transport of waste and the transport and application of compost as well as the benefits from compost application are equal as in the scenario of anaerobic digestion.

CDM Results

Scenario 0: Table 1 shows the total and yearly average emission reductions that could be achieved by avoiding waste disposal, calculated for different crediting periods and 1 t organic waste per year landfilled. The table also shows the variation in results depending on the climate zones, which are characterized by temperature and dryness (influencing the decay rates used).

Table 1: Total and yearly achievable emission reduction by avoiding landfill disposal, in kgCO$_2$e for 1 t organic waste per year for different climate characteristics (Volkart, 2011).

Crediting Period	Total				Yearly Average			
	≤20°C		>20°C		≤20°C		>20°C	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
7	993	1996	1281	2923	142	285	183	418
10	1859	3513	2359	4831	186	351	236	483
21	6609	10’543	7996	12’728	315	502	381	606

Scenario 1 and 2: The results for project emissions of anaerobic digestion and for the two different composting technologies are shown in Figure 2. The calculations indicate that the largest share of emissions is caused by on-site fuel and electricity consumption and CH_4 emissions from degradation

of organic matter or from leakage of the biogas plant. The CH4 emissions from composting account for 84.00 $kgCO_2e/t$ and are slightly higher than the estimated physical leakage of 69.73 $kgCO_2e/t$ from anaerobic digestion.

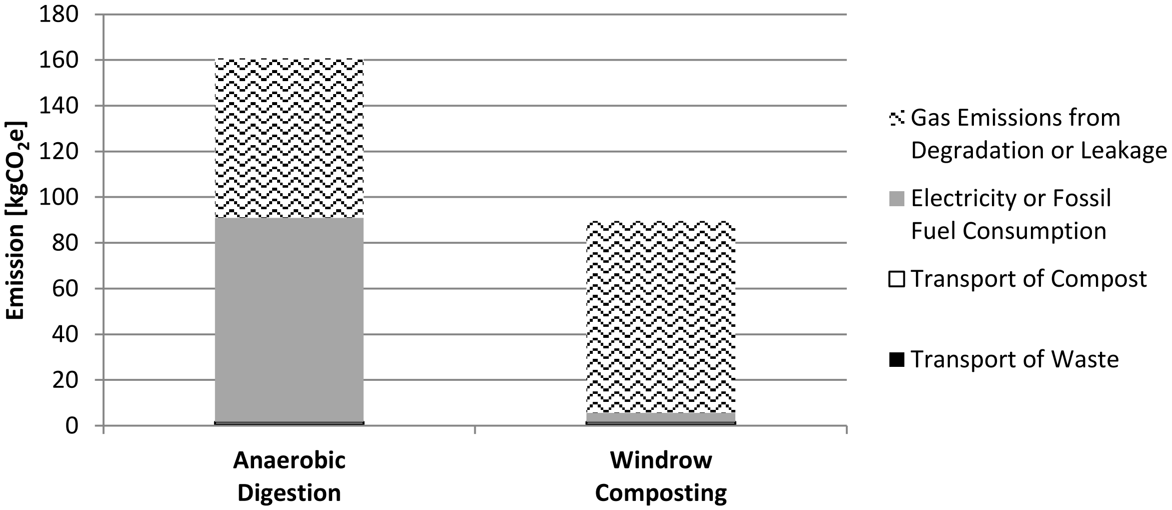

Figure 2: Project emission by categories for anaerobic composting (scenario 1) and windrow composting (scenario 2) in $kgCO_2e$ per ton of organic waste and year (Volkart, 2011).

The biogas plant accounts for the emission of 89.24 $kgCO_2e/t$ from power consumption whereas the emissions from energy consumption from the windrow compost facilities are much lower. Besides the avoidable emissions of landfill disposal, anaerobic digestion will also avoid emission through substitution of energy as well as avoiding emissions from wastewater treatment (as the plant uses wastewater as co-substrate) . When using the gas to produce electricity 133.45 $kgCO_2e/t$ are avoided. With gas use as thermal source the emission reductions are slightly lower (119.35 $kgCO_2e/t$). Avoiding emissions from wastewater treatment accounts for 78.87 $kgCO_2e/t$.

Sensitivity Analysis

With the LCA approach various uncertainties were assumed and the respective sensitivities calculated. For landfill disposal calculations were also conducted for an oxidation rate in the soil cover of zero (instead of 25%). This causes an increase of 44.01% considering GWP. Similar results were calculated for landfill gas flaring, where a non-functional landfill gas capture and flaring system would increase GWP by 34.79%.

High uncertainties exist in the available data considering the greenhouse gas emissions during the decomposition of organic waste by anaerobic digestion or composting. The total GWP from anaerobic digestion ranges from <1% up to 137% of the default scenario, depending on the factors used for emissions caused during the degradation processes of waste or digestate before and after the anaerobic treatment. If additionally also different leakage factors are considered, the GWP may vary up to 274% of the default result. The compost plant shows a smaller variability ranging from 48% to 192% of the calculated default result of GWP. Using the default emission factor from IPCC (2006) leads to the highest value. The influence of other uncertainties were also calculated in Volkart (2011) - such as: i) choice of composting technology (forced aeration or closed reactor composting); ii) variation of emissions from the application of compost/biodigest; iii) variation of the use of gas; iv) changing of transportation system - but are all not further reported in this paper.

COMPARISON OF RESULTS AND CONCLUSIONS

In comparison with the results from LCA, the emission reductions quantified with the approved CDM methodologies for composting and anaerobic digestion seem to underestimate the real emission reduction potential of the project activities. Already the calculations of baseline emissions differ. However these results have to be interpreted with caution as CDM and LCA use different approaches.

While LCA assesses the emissions of a certain amount (1 t) of organic waste over a period of 100 years, the CDM Baseline Methodology calculates the emissions over a certain time frame (usually per year) based on the crediting period (7, 10 or 21 years). For short crediting periods the potential to avoid emissions is smaller and the difference to the result from LCA increases. The impact of climate is also reflected in the register of already validated composting projects. All of the 41 validated small scale projects introducing composting of organic waste were carried out in rather tropical climates of Malaysia, Indonesia, India and Brazil (UNFCCC, 2011). Considering the climate in Cochabamba, the decay rates to be used would be rather low, hence only a small share of waste would be degraded after the end of the crediting period resulting in low amount of emission reduction credits.

Another difference is in the conversion factor of CH_4 to CO_2e. The indicator IPCC 2007 GWP 100y - as used in the LCA - considers the factor for CH_4 of 25 CO_2e (Forster et al., 2007) based on the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. However, the Kyoto Protocol was signed in 1997 while the Second Assessment Report of the Intergovernmental Panel on Climate Change was still state of the art. This report suggested a factor for CH_4 of 21 CO_2e (Houghton et al., 1995). Hence the result from CH_4 emissions assessed with the CDM methodology will always be less than when using IPCC 2007 GWP 100y.

Emissions caused by the application of compost are also treated differently in the two methods. In the LCA, the application of compost has a positive impact while the CDM methodology calculates an overall negative impact. This is because the CDM methodology only takes transport emissions from compost into account and does not consider any further emissions or emission reductions such as the substitution of inorganic fertilizer as it is included in the LCA.

In CDM methodologies only greenhouse gas reduction potential of a project is assessed while all other environmental impacts are neglected. For waste management projects such as composting and anaerobic digestion, this seems reasonable as the majority of the environmental impacts are determined by greenhouse gas emissions. However, the sensitivity analysis showed that the system is also sensitive to further impacts such as fossil fuel depletion, particulate matter formation (NH_3 and NO_x emissions) or heavy metal inputs by application of compost. Toxic emissions might even overcompensate the overall beneficial impact from GWP reduction. Here, wider consideration of additional impact factors can create incentives to develop projects that reduce greenhouse gas emission, but additionally also ensure reduction of other environmental impacts.

Generally, large uncertainties exist in the knowledge about greenhouse gas emissions from processes such as composting and anaerobic digestion as well as the application of the produced fertilizer. Because of the sensitivity of these factors more research should be carried out in this area in order to improve the quantification of the amount of gases emitted as well as achieving more evidence on the influence of the specific treatment practices.

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Bioreactor Landfills: An Answer to Disposal of Msw for Developing Countries

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ABSTRACT

Municipal Solid Waste (MSW) generation rate as well as quantity is increasing everyday due to increasing population and urbanisation especially for developing countries. In major part of India, MSW is simply dumped on a wasteland at the outskirts of the city, making lives of the residents in the vicinity miserable.

The Sanitary Landfill for disposal of MSW takes too much time for waste stabilization on account of which it becomes a gigantic tasks to safely and scientifically dispose MSW.

Bioreactor Landfills though a relatively new concept in India could be an answer to this problem. To study the MSW Stabilization in Bioreactor landfill, a Lab Scale Pilot Bioreactor was designed and operated to simulate Bioreactor landfill conditions. With the help of this study as well as work done on Bioreactor landfills earlier, it has been attempted to technically and economically compare Bioreactor Landfills to Sanitary landfills.

INTRODUCTION

MSW is a collection of all solid unwanted things from a Municipal (urban) area[1]. It includes a varied assortment of items like refuse, construction and demolition waste, sludge, leaves and bulky items. As per the census data of 2011, current Indian urban population is around 28 percent of total population, i.e. around 338 million. Considering an average per capita waste generation of 0.6 kg/day, waste generated daily amounts to 0.202 million tonnes. If the whole MSW is properly managed and disposed, then it has a potential of providing a reasonable quantity of alternate renewable energy apart from the above discussed recyclables and hygienic conditions. However, as per the above mentioned survey only 8 percent of the MSW is scientifically disposed (in sanitary landfills), thus the balance MSW which is merely dumped on the outskirts of the urban area is a loss in terms of energy derived from MSW. Talking about MSW from Ahmedabad, one tonne of MSW can generate 27.3 cu. m. of methane at STP(calculated based on data provided by Ahmdabad Municipal Corporation)[1]. Assuming the same amount of methane potential for the MSW for the whole country,methane that can be generated from the MSW generated from the whole country amounts to 5.51 million cu. m. per day. However, this methane potential can only be achieved in a reasonable time if all of the MSW is disposed of in Bioreactor Landfills.

A bioreactor landfill is a sanitary landfill that uses enhanced microbiological processes to transform and stabilize the readily and moderately decomposable organic waste constituents within 5 to 10 years of bioreactor process implementation.[2][4][9][11] (For highly decomposible waste, decomposition can be completed within days). The process requires significant liquid addition through leachate recirculation (single most important strategy) to reach and maintain optimal conditions[9]. Strategies including waste shredding, pH adjustment, nutrient addition, waste pre-disposal and post-disposal conditioning, and temperature management, may also serve to optimize the bioreactor process[9].

The bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates, stabilization and process effectiveness over what would otherwise occur within the landfill. Stabilization means that the environmental performance measurement parameters (landfill gas composition and generation rate and leachate constituent concentrations) remain at steady levels, and should not increase in the event of any partial containment system failures beyond 5 to 10 years of bioreactor process implementation.

COMPONENTS OF BIOREACTOR LANDFILLS

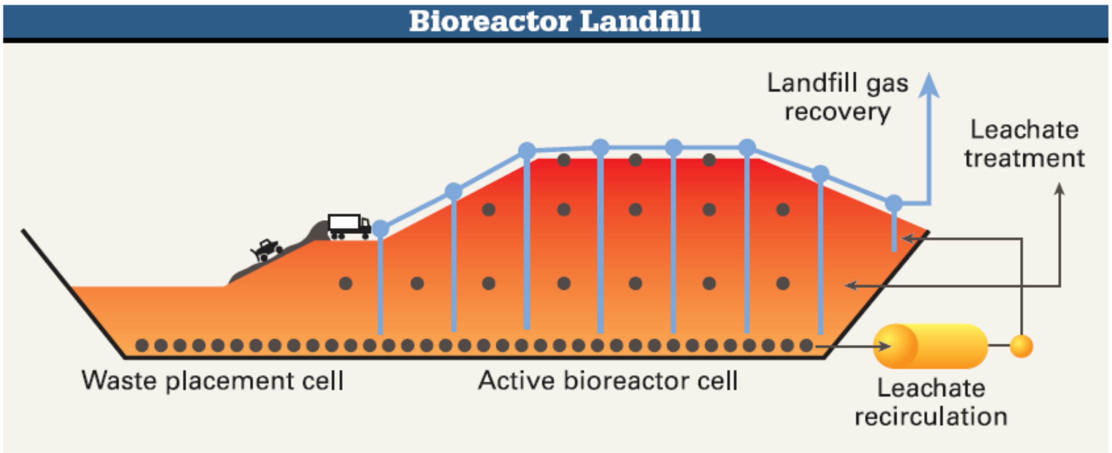

Fig.-1 Sketch of Bioreactor landfill showing all components (Source: Patrick Walsh and Philip O' Leary (2002))

The seven essential components of a landfill are[2]:
(a) A liner system at the base and sides of the landfill
(b) A leachate collection and control facility
(c) A gas collection and control facility (optional for small landfills)
(d) A final cover system at the top of the landfill
(e) A surface water drainage system
(f) An environmental monitoring system
(g) A closure and post-closure plan

PROCESS MECHANISM IN BIOREACTOR LANDFILLS

Numerous studies have been carried on the anaerobic biodegradation process in the landfills. Some investigations have suggested that the stabilization of waste proceeds in five sequential and distinct phase. The rate and characteristics of produced leachate and biogas vary from one phase to another, and reflect the microbially mediated processes taking place inside the landfill. The phases are as under[1][9]:-

Phase I: Initial Adjustment Phase

In the aerobic phase both oxygen and nitrate are consumed with soluble sugars serving as the carbon source for microbial activity. The quantity of oxygen available is fairly low, depending on the degree to which the waste is compacted. The initial phase is associated with initial placement of solid waste and accumulation of moisture within landfills. An acclimation period (or initial lag time) is observed until sufficient moisture is develops and supports active microbial community.

Phase II: Transition Phase

With the depletion of oxygen trapped within a landfill, a transformation from an aerobic to anaerobic environment occurs, and the facultative anaerobic microorganisms become active. The electron acceptors shift from oxygen to nitrates and sulfates. The hydrolytic and fermentative microorganisms hydrolyze polymers such as carbohydrates, fats and proteins. By the end of this phase, measurable concentrations of COD and volatile organic acids can be detected in the leachate. In addition, the ammonia can be detected due to the hydrolysis and fermentation of protein compounds.

Phase III: Acid Formation Phase

During the first stage of this phase, the intermediates produced from phase II, such as sugars, amino acids, etc are further fermented into short-chain carboxylic acids, carbon dioxide and hydrogen. Acetate and alcohols are also formed. During the second stage of this phase, the obligate proton-reducing acetogens become active. They oxidize the fermentation products of the first stage to acetate, carbon dioxide and hydrogen. There is a hydrogen scavenging population i.e. methanogens in an active anaerobic ecosystem. If fermentative and methanogenic activities are not balanced, intermediates will accumulate and may percolate from the landfill as leachate. Therefore, intermediate

VOAs at high concentrations and a decrease in pH accompanied by metal species mobilization are often observed before the onset of MSW methanogenesis. The viable biomass growth associated with the acid formers bacteria, and rapid consumption of substrate and nutrients are the predominant features of this phase.

Phase IV: Methane Fermentation Phase

During this phase, both methanogens and sulphate reducing bacteria are involved in the anaerobic degradation. Methane gas constitutes approximately 50-60% (by volume) of gas composition. The pH value is increased, and consequently heavy metals are removed by precipitation. The organic matter present in the leachate declines which causes the BOD and COD to fall. In the mean time, sulphate-reducing bacteria convert hydrogen, acetic acid and higher volatile fatty acids into carbon dioxide and hydrogen sulphide. This group of bacteria competes with the methanogenic bacteria to transform the hydrogen and organic carbon.

Phase V: Maturation Phase

The easily biodegradable organic matter is stabilized, and nutrients and available substrate become limiting. Gas production drops dramatically and leachate strength stays steady at much lower concentrations. Reappearance of oxygen and oxidized species may be observed slowly. The low level biodegradable matter gradually humifies.

DESCRIPTION OF PILOT BIOREACTOR

The components of Bioreactor landfills are already mentioned in the earlier section. In order to simulate a Bioreactor landfill conditions in laboratory to test variations of parameters, a set-up known as Lab scale Pilot Bioreactor was created. It consisted of a PVC Cylindrical drum of diameter 40 cms and height 80cms was used as bioreactor. The surface of the PVC drum served as liner material, preventing any leachate from flowing outsided it. The bottom 15 cms depth of the drum was filled with Marble Chips, which acted as leachate collection layer. A hole at the top of this layer was made at one side of the PVC drum in which a 12.5 mm diameter pipe was inserted to collect the leachate accumulated in this layer and carry it to a bucket placed just outside the drum.

Fig.-2 photo of lab scale pilot bioreactor

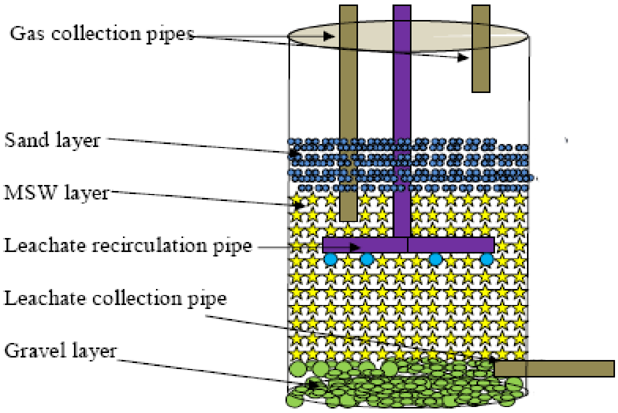

Fig.-3 L- section of lab scale pilot bioreactor

Above the gravel layer (of Marble chips) was placed a layer of MSW 45 cms depth. Above the MSW layer was placed a 10 cms layer of sand which acted a layer of cover system. The cylindrical drum was covered with a flex sheet so as to create anaerobic conditions inside the drum or reactor. From the flex sheet holes were made to insert a leachate recirculating pipe going till the MSW layer. Also two pipes were inserted to collect LFG, if generated. One of the pipe extended till the MSW layer, while the other pipe ended just above the sand layer. Both the pipes contained holes so as to trap LFG. The top of the two pipes were covered tightly by polythene bags so as to trap the gas generated in the reactor. Care was taken to seal the junction of flex sheet and the three pipes so as to prevent leakage of gas. After recirculation of water through the leachate recirculating pipe, the top end of the pipe was sealed with polythene bag to prevent leakage of gas generated inside the reactor through this pipe.

DESIGN

The reactor was designed in such a way to simulate Bioreactor landfill conditions as closely as possible, considering the limitations of Pilot studies. Simple engineering design was used in constructing and operating the reactor.The height of the test reactor or PVC drum used is 80 cms and diameter 40 cms. As discussed earlier, the bottom 15 cms of the reactor consists of gravel (marble chips). Leaving an empty space of 10 cms at top, and 10 cms for sand layer below the empty space, the available height for MSW cum cow dung layer is 45 cms. Since, the diameter of the cylindrical drum reactor is 40 cms, available volume for MSW cum cowdung layer is 0.056 cu.m. Considering the bulk density of MSW as 0.90 tonnes per cu.m., The weight of MSW in the available volume comes out to be approximately 50 kgs. 4 kgs of fresh cow dung was added to the MSW, thus making the weight of the waste used in the reactor as 54 kgs.

Calculation Of Leachate Quantity To Be Recirculated.

Assuming initial moisture content of MSW to be between 15% of saturation moisture content and targeting a moisture content of 40% so as to attain field capacity [8], the volume of water required shall be approximately 21 litres.

THE OPERATING PARAMETERS

For finding out the effectiveness of Bioreactor Landfills and Processes therein, following parameters were evaluated[6]:-

- Leachate Collected.
- Periodic COD of the Leachate.
- pH variations over the period.

These parameters were considered for both the reactors and their interrelationships were also studied inorder to arrive at the conclusions.

METHODOLOGY

After reading up the designed set-up, 21 litres of water was circulated on the first day. The leachate collected from the liquid addition on the first day was mixed with fresh water was added to it so as to make the volume of leachate equal to 21 litres. Same procedure was repeated for the whole period of study. Weekly test of pH and COD of leachate was done during the period of study and the results are shown in the graphs in the next section.

DISCUSSION OF RESULTS

Observations of the study depicted above point to certain conclusions discussed below. Separate conclusions based on different parameters have been made for the ease of understanding.

Leachate Collected

Approximately 85 percent of leachate recirculated was collected in 24 hours of leachate recirculation over the period of the study. Initially more than 85 percent of leachate recirculated was collected in 24 hours which got reduced progressively as shown in the fig.-4. However, still the overall variation in the leachate collection during the period of study was not appreciable.

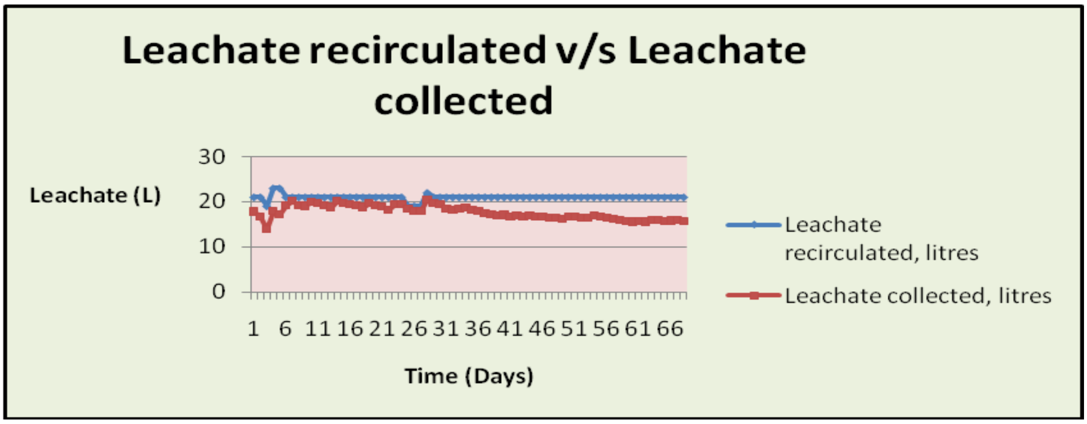

Fig.-4 Graph of leachate recirculated v/s leachate collected

Chemical Oxygen Demand.

COD of leachate from the lab. scale pilot bioreactor varied considerably (fig.-5). Initially the COD was in the range of 6000-7000 mg/L for first few days, with progressive increase with time. It reached its peak value of 14000 mg/L after 20 days of leachate recirculation. After that the COD started decreasing at appreciable rates initially but steadily at lower rates after one and half months of the operation of the reactor. The COD of the leachate was around 8000 mg/l at the end of two months of the reactor operation, at the end of the period of study. The reason for initial increase in COD of leachate could be due to conversion of more and more particulate COD into soluble COD and its dissolution into the leachate recirculated. However, with time, (after 20 days) the COD started decreasing, due to microbial activity resulting into stabilization.

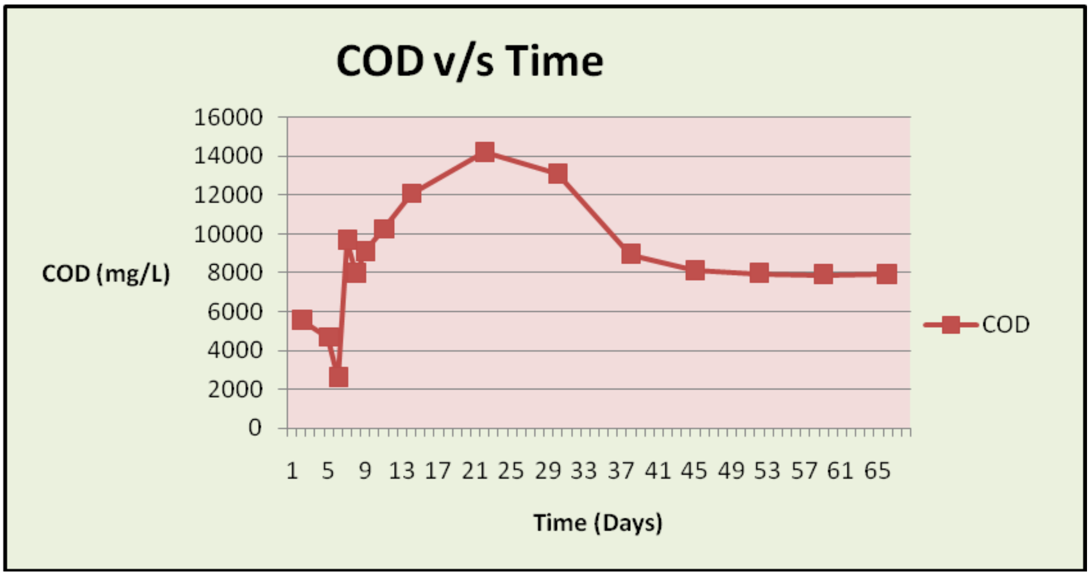

Fig.-5 Graph of COD v/s time

pH

pH of the leachate was highly acidic at the start of the study. It was around 3.1 initially and with time it showed progressive increase, reaching upto pH 7 at the end reactor operation (fig.-6).

It is to be noted that with the progression of the study the quantity of leachate collected went on decreasing. This could be due to the possible compaction of the bed of waste.

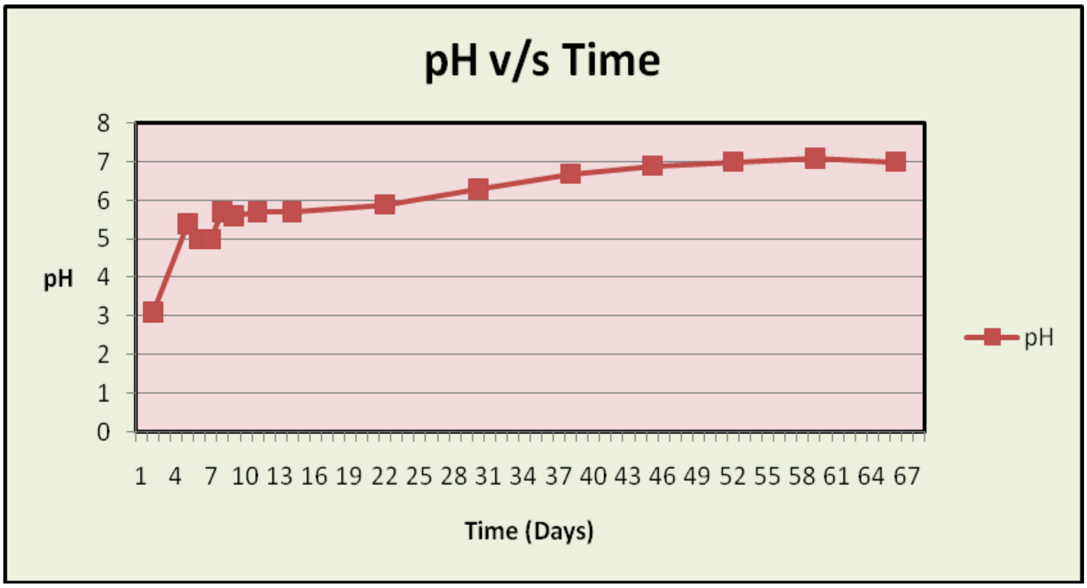

Fig.-6 Graph of pH v/s time

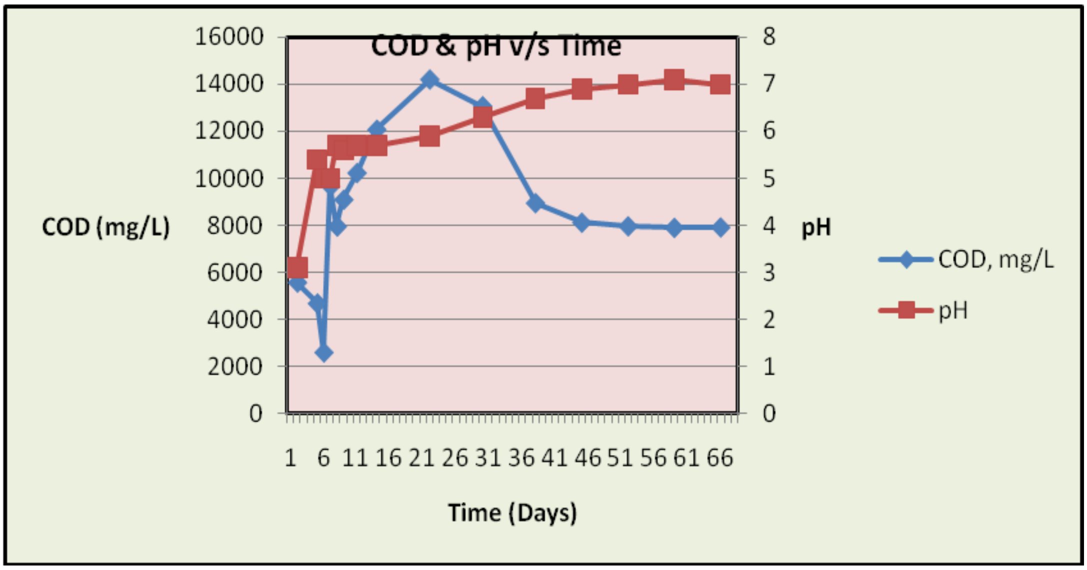

Fig.-7 Graph of COD & pH v/s time

ECONOMIC ANALYSIS

Cost Of Construction

Making a economic comparison between a Sanitary Landfill and Bioreactor landfill is a tremendous task, considering no existence of landfills in the study area or even in the state of Gujarat. However, best efforts have been made so as to arrive at an economic comparison. For this Central Public Health and Environmental Engineering Organisation (CPHEEO) manual on Solid Waste Management is referred by the author, where in costs of various items for constructing a sanitary landfills is given. The costs or prices correspond to 1998 prices. However to arrive at the latest prices, that is to say 2011 prices, use of Consumer Price Index for Urban Non Manual employees was done. Besides that, reference to current market prices for certain items was also made so as to be as accurate as possible in arriving at the economic comparison.

As per the Central Public Health and Environmental Engineering Organisation, Cost Analysis for a 1000 tons per day Sanitary Landfill was given based on the assumptions as under[5]:-

Table-1 Assumptions for Construction of Landfill

Sr. No.	Criteria	Description
1	**Waste Generation**	1000 tons per day
2	**Design Life**	Active Period 16 years Closure and Post Closure Period = 25 years
3	**Topography**	Flat Ground
4	**Subsoil**	Sandy silt upto 20m below ground surface, underlain by bedrock
5	**Water Table**	10 m below ground surface
6	**Average Total Precipitation**	750 mm per year
7	**Base Year**	1998

Besides above data, certain assumptions are made in the design and based on the calculations and above assumptions, the cost analysis for conventional sanitary landfill and Bioreactor landfill is done as under:-

Sanitary landfills

Table-2 Cost of Construction for Sanitary Landfill

Sr. No.	Item	Average Cost (Rs)	No of times the cost is considered	Total Cost Rs
1	Site Selection and Site Characteristics	6.01 million	1	6.01 million
2	Design and Detailed Engineering Cost	3.91 million	1	3.91 million
3	Site Development Cost (Excluding land acquisition)	35.81 million	1	35.81 million
4	Phase Development Cost (Yearly)	95.44 million	16	1527.16 million
5	Phase Operation Cost(Yearly)	36.83 million	16	589.409 million
6	Phase Closure Cost(Yearly)	39.34 million	30	1180.27 million
7	Post Closure Care Cost (Yearly)	8.27 million	30	248.19 million
TOTAL COST (Rs)				3590.77 million

Note: Latest Cost of Items 1-7 have been arrived at using Consumer Price Index (for urban non manual employees), which is the closest possible index to arrive at current prices. However, it should not be considered an accurate determination for the latest prices. The prices determined using the Index are much on the lower side and for all practical purposes detailed estimation based on prevailing market prices should be used.

Bioreactor landfills

According to the Literature available and partly as per this study, it is clear that Bioreactor landfills have a potential to stabilize the waste in far lesser time as compared to conventional sanitary landfills. To be specific, let us consider bioreactor landfills stabilize a given quantity of Municipal Solid Waste in 10 years (As per the available literature), the time taken by Sanitary Landfills for stabilizing the same quantity of MSW could be between 30 years to 100 years. Being specific, it can be taken as 30 years,with proper operation of a Sanitary Landfill.

Hence, Bioreactor Landfills stabilize 3 times the waste stabilized by Sanitary landfills. The cost of operation of Bioreactor landfill is as under:-

Table-3 Cost of Construction of a Bioreactor Landfill

Sr. No.	Item	Average Cost (Rs)	No of times the cost is considered	Total Cost Rs
1	Site Selection and Site Characteristics	6.01 million	1	6.01million
2	Design and Detailed Engineering Cost	3.91 million	1	3.91 million
3	Site Development Cost (Excluding land acquisition)	35.81 million	1	35.81 million
4	Phase Development Cost (Yearly)	95.44 million	16	1527.16 million
5	Phase Operation Cost(Yearly)	36.83 million	16	589.40 million
6	Phase Closure Cost(Yearly)	39.34 million	10	393.42 million
7	Post Closure Care Cost (Yearly)	8.27 million	10	82.72 million
8	Leachate Recirculation cost (Yearly)	85.31 million	10	853.14 million
TOTAL COST (Rs)				349,16,08,588

For Bioreactor landfills, all the above costs apply. In addition, costs of Leachate recirculation also comes into picture. The costs of leachate recirculation for the above scenario comes out to be Rs 85.31 million (Yearly). (The Pipes considered for Leachate Recirculation are NP2 and PVC pipes.). There are other costs incurred as well like for pH maintenance, nutrient addition, etc. But they are insignificant as compared to leachate recirculation costs and hence are not considered here.

Thus Total benefits accruing in monetary terms in using Bioreactor landfills is Rs 99.16 million. In addition saving of atleast 20 years of Post closure operations is achieved. Besides the same land can be reused thrice in the time interval of 30 years by putting bioreactor landfills to operation as against sanitary landfills.

MONETARY BENEFITS FROM METHANE CAPTURE

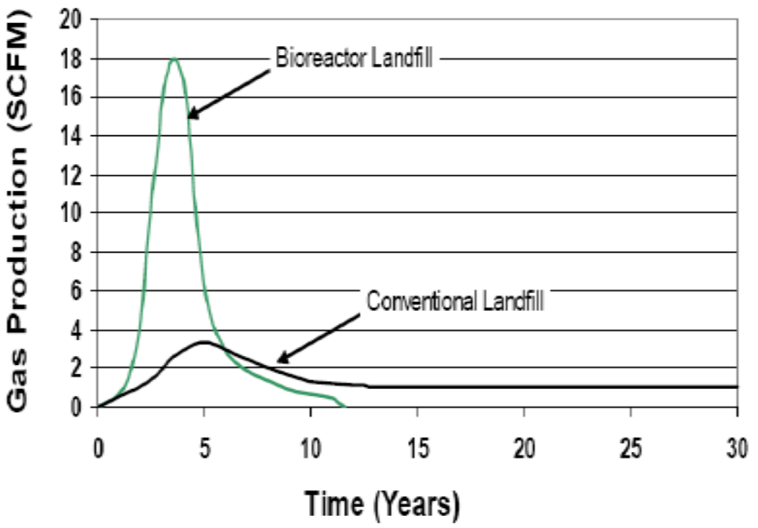

Fig.-8 Graph showing comparison of gas produced v/s from Sanitary landfill and Bioreactor landfills (source:- 2007 Yolo County Yazdani)

80 percent of methane generation in case of Bioreactor landfills takes place within 4-5 years. At standard temperature and pressure conditions, Methane produced in 4 years from bioreactor landfills is about 10,12,072 cu.m. considering the calorific value of methane as 39,820 KJ/cu.m., the energy obtainable from the methane generated from bioreactor landfills within 4 years is 4030,07,07,040 KJ or 27603224 KJ/day or 11,50,134 KJ/hour.

During the same period, methane produced from sanitary landfill is only 1,78,601 cu.m. or 711,18,91,119 KJ or 2,02,965 KJ/hour. Thus the energy obtained from bioreactor landfills is about 5 times higher than in sanitary landfills, which translates into 5 times more earnings through the sale of energy. Besides, since 80 percent of methane is captured within 4-5 years in case of bioreactor landfills, the capturing methane becomes an attractive proposition, unlike in sanitary landfills, wherein methane potential is realised in more than 30 years[10].

CONCLUSION

Bioreactor Landfills though have existed in many countries for more than 2 decades are new for India as no such landfill exist in India as of now. This author has tried to highlight the technical and economic benefits of Bioreactor landfills as compared to Sanitary Landfills and it is evident that on both counts Bioreactor Landfills give much better results. Bioreactor landfills have the potential to stabilize the MSW in one third of the time or even faster especially in Indian conditions as compared to Sanitary Landfills. Besides that, on economic front too, Bioreactor landfills lead to decent savings especially due to higher methane generation in shorter period.

However, with Bioreactor landfills, leachate recirculation needs to done apart from controlling other parameters, which require careful monitoring as well as proper implementation of process, which at times could be troublesome, but worth doing also. Another drawback with Bioreactor landfills is emission of pungent odour with leachate recirculation, which needs to controlled in order to avoid nuissance to nearby residents and the people working at the landfill site.

To Conclude, the benefits far outweigh the costs and bioreactor landfills could be adopted for Indian conditions albeit gradually as lot of research needs to be done so as to find out best combination and proportion of co-substrate, leachate recirculation requirements, nutrients required, etc.

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Process optimization strategies – The utilization from stakeholders experience of biogas plants

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ABSTRACT

The research approach for process optimization from biogas plants range from new measuring devices to complex mathematically models. Contrary this trend the presented method integrates conventional process control strategies and the experiential knowledge from stakeholder which can influence the biogas process. Therefore, artificial intelligence (AI) and deterministic models in environmental issues are presented to figure out the potential for the use as a strategy for biogas plant process optimization strategies. Beside this, a stakeholder analysis is realized to figure out the responsible persons for process optimization strategies from biogas plants. The AI method case-base reasoning (CBR) is described more in detail to integrate the decision making process from involved people regarding input material specification and process monitoring. The organoleptic examination, the description of organic input properties by the use of human senses is presented more in detail.

INTRODUCTION

The use of renewable energies is the main interest of environment protection regarding the growing energy consumption and the reduction from fossil fuels resources. One strategy is the anaerobic digestion from organic matter e.g. organic waste, agriculture residues, waste water sludge and renewable energy crops. In Germany more than 7.500 agriculture biogas plants are installed. Beside the impact of the green house effect, the main goal for agriculture biogas plant operators is profit maximization. To achieve the financial goal the risks of the technical a biological process has to minimized, the reduction of costly input materials and the efficient use of them is necessary and the energy production has to be increased. All this factors are obtainable with process optimization.

During the recent years, a rising complexity in measuring methods and process control systems at biogas plants can be observed in Europe. On the one hand more and more measuring devices are available for biogas plants e.g. flow meter, online micro wave measurement, near-infrared-spectroscopy (NIRS). One the other hand many complex process control systems exists for lab scale fermenter units, e.g. based on the ADM 1 or fuzzy logic. The combination of complex model based control units with a simulation support and many input parameters is a research topic for a large usual biogas plants. Therefore, many challenges for research facilities exist.
Against this trend the utilization of more and complex measurements and simulations, experts developed monitoring and diagnosis systems which based on an expert system in the field of waste water treatment plants. They use the knowledge of stakeholder regarding the specific process to transfer their decision making into technical language and operations. This knowledge management system helps to operate with complex situation and many impact factors.

In the context of anaerobic digestion in biogas plants, a great potential in the use of expert systems (XPS) exist. The assessment of biological processes and the calibration of sensors are still running with empirical knowledge. Regarding limit values of biological processes valid experiments and the interpretation from experts are necessary. In terms of increased efficiency from biological treatment of waste, technical sensors and the empirical knowledge of operators is useful to combine.

This paper gives an overview about the use of artificial intelligence (AI) methods to optimize the biogas plant. With case-based reasoning (CBR) a method to include experiential knowledge from biogas plant stakeholder is available. Therefore, a stakeholder analysis and the capture of the experience from biogas stakeholder are presented. With the use of the organoleptic examination – a method to use the human senses – and the AI method CBR, one aspect of a new knowledge based control system for biogas plants is presented.

STAKEHOLDER ANALYSIS FOR BIOGAS PLANT OPTIMIZATION

Optimization strategies are influenced from the point of view – the balance frame. One point of view is the plant frame which process steps, measuring devises or technical equipment are integrated. Another point of view is the stakeholder analysis. The interests hereby are which people are involved and which interests they follow.

Regarding the stakeholder analysis for the optimization from biogas plants following structure (Figure 1) shows an overview. The centre describes the mission: process optimization. How close a stakeholder belongs to this mission indicate how strong their interests are for a successful operation. Stakeholders are distinguishable into internally, primary and secondary stakeholder.

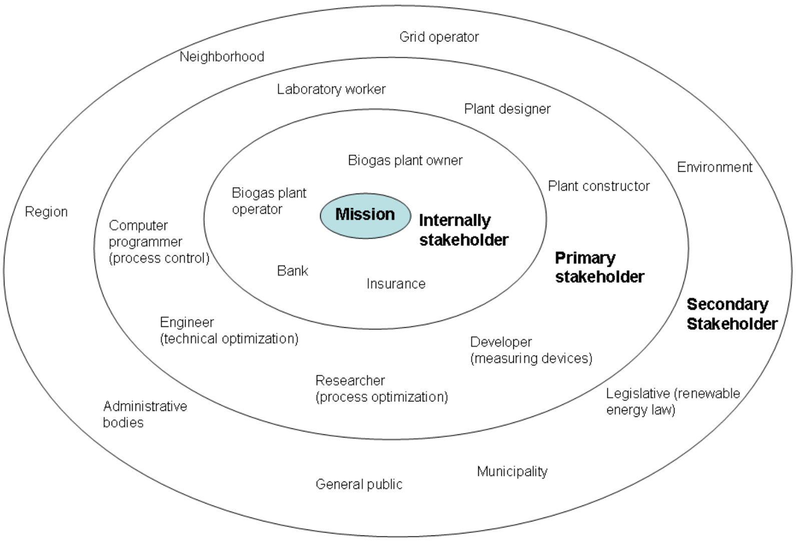

Figure 1 Stakeholder analysis for process optimization strategies from biogas plants

With the help of this analysis method internally stakeholders and their behaviours and influences are describable. Following stakeholders are identified:

- Biogas plant operators and owners
- Bank and insurance.

The interests in plant optimization are different and go from a high stakeholder value until risk minimization. Both, biogas operator and biogas owner are mainly focused on a biological steady state of the biogas process. Biogas operators are interested in the functionality from the engines, reactor and the general technical equipment and measuring devices. They are focused on the MTBF, the mean time between failures. Every change from a machine or some special mechanical or electronic components need working time and new invest. Therefore, a minimization from errors and machine failure and a good running biogas process are the main interests of the biogas operator regarding process optimization strategies. Another stakeholder is the biogas plants owner. They are interested in a high stakeholder value that means a high amount of profit with low-risk. Therefore, the optimization of the use from input materials, the biological process optimization and the uninterrupted generation from biogas is necessary.

The focus from banks is the risk minimization from their lending, a credit portfolio with the lowest possible risk. This is possible through a good analysis from the biogas plant regarding the technical equipment the material input and the calculated gas amount. On the other hand a better understanding about the anaerobic digestion and their disturbances is useful. Are there so many unanswered questions the use of dear interests is the risk minimization from the bank. The insurance companies working with validations and regulation options as well.

Beside this identified internally stakeholders a good planning, construction and operation from the biogas plant is necessary. Regarding the process optimization the use of measurement and control technology is inevitable. Researcher and biogas plant operators have a lot of experience regarding the anaerobic digestion process and this knowledge is useful and adoptable in the process control system as well.

Artificial intelligence and deterministic models in environmental issues

Generally aspects about the artificial intelligence (AI) methods in the field of environmental issues CORTÉS ET AL. (2000) gives an overview about strategies, methodologies and applications for technical optimization of environmental processes. The clarification of the main steps to establish an decision support system (DSS) are – data interpretation, data mining and problem diagnosis –. This cycle of new input, validation and storage is the basement of the implementation from experiential knowledge from process operators or observers. Among the AI methods often used in the developed DSS in the past years CORTÉS ET AL. (2000) noting:

- Rule-based reasoning,
- Model-based reasoning,
- Fuzzy logic techniques and
- Case-based reasoning.

On the basis of a rule-based reasoning expert system PUÑAL ET AL. (2001) established a monitoring and diagnosis model for anaerobic wastewater treatment plants. The use of monitored variables with the help of online measurements and the integration of offline parameters is combined. The process description as a process state is defined by the connection of the methane concentration, the gas flow rate and the feed flow rate. By the use of mathematically operators like IF, AND and THEN the architecture of this diagnosis system is described and expert based recommendations are available. The diagnosis from the experts during the laboratory experiments and the summary of them together with the measured variables make it possible to collect defined cases with defined recommendations to optimize the process. This procedure allows establishing a computer based monitoring and diagnosis system for anaerobic wastewater treatment plants in the laboratory scale. The transfer to a biogas plant with the feedstock biowaste and agriculture residues is possible with some adoptions and based on condition monitoring with the use of the experience of plant operators.

The use of fuzzy logic as a knowledge-based methodology in the field of AI in environmental issues is established. The description form complex biological processes and long term parameter alteration required an adequate measurement set and a process control system. As a model based system Anaerobic Model Nr. 1 (ADM1) is available with the input of a lot of variables. This model-based reasoning is running in the lab scale for wastewater treatment plants as well as biogas plants [CIMATORIBUS 2009]. However, the installation on large scale biogas plants doesn't exist in a long term measurement campaign. Therefore, the use of the fuzzy logic methodology with the utilization of a small number of input parameters and the integration of expert knowledge for interpretation of the process state is common and applicable for large scale treatment plants.

The integration of expert knowledge trough the connection of variables with fuzzy logic techniques NUMERS ET AL. (1994) published a fundamental diagnosis system for the control of specific bioprocesses (production of lactic acid). This methodology allows the reasoning process of a human operator regarding the use of a knowledge-based system (KBS). The implementation of new experiences is possible cause the validation of new data sets, control strategies and the recommendations from the operator. One year before AYNSLEY ET AL. (1993) presented a knowledge-based system by neural networks. This method allows integrating non-linear and time variant bioprocesses in the case of penicillin production and the relevant expertise and experience from engineers, plant operators and researcher. The lessons learned are: better sensors, automatic analysis systems and knowledge-based systems for bioprocess scheduling operations.

Nowadays, AI methods and the better knowledge of the anaerobic process steps and the computing capacity allow establishing knowledge-based support system for process diagnosis and for simulation strategies. The research topic of human-computer-interaction combines technical possibilities and the integration of human expert knowledge to a new science.

The last AI method which described CORTÉS ET AL. (2000) as a knowledge-based method for environmental issues is case-based reasoning (CBR). In the field of organic treatment WIESE ET AL. (2005) present a combination or real-time control (RTC) and the use of CBR for a decision support system (DSS) for wastewater treatment systems. This optimization strategy integrates a case-based predictive controller which simplified the utilization of sequencing batch reactors (SBR) with a low measurement data set.

The use of this machine learning techniques withstands the requirement for an online-based computer system. Therefore, combinations of case-based reasoning and rule-based reasoning [SÀNCHEZ ET AL. 1995], CBR and fuzzy logic techniques [BOUCHON-MEUNIER ET AL. 2009, WATSON 1999] exist in field of AI of environmental issues.

Beside this presented methods for knowledge-based control systems. VINDIS ET AL. 2009 developed a multi-criteria assessment of energy crops for biogas production by the use of hierarchical decision trees and the numerical combination of the used elements. The idea is to implement much more criteria which describe the situation than online measurement data or expert diagnosis. This method allows the use of soft parameter and numerically measured attributes for decision support systems.

With the help of case-based reasoning (CBR) as a basis for an expert system with individual decisions are describing more precisely in a way of a machine learning AI technique. AAMODT ET PLAZA (1994) presented foundational issues, methodological variations and system approaches of CBR. The idea of CBR is the problem solving paradigm by the use of related problems and their solutions regarding the new case. Figure 2 shows the case-based reasoning methodology concerning the cases which implement a problem and the specific solution. The second step is the comparison of a new problem with the cases to find an analogy. The goal is to find a parallelism from stored cased to the new one to purpose an adapted solution.

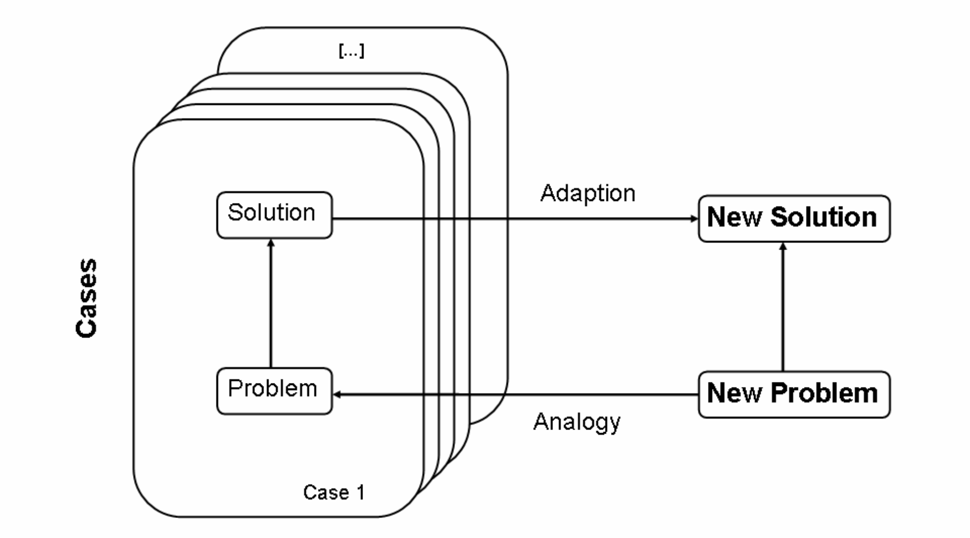

Figure 2 Case-based reasoning methodology [modified, AAMODT ET PLAZA 1994]

This decision support system is able to learn cause the validation of the new problem and the new solution. If this new case is acceptable, the new case is added into the cases. This methodology is similar to neural network but differs in transference of the decision-making process. WATSON (1999) described the CBR-cycle as follow:

- Retrieve similar cases to the problem description,
- Reuse a solution suggested by a similar case,
- Revise or adapt that solution to better fit the new problem if necessary and
- Retain the new solution once it has been confirmed or validated.

This cycle requires the search for the nearest neighbour regarding the problem description with attributes and the value of them. With the similarity between a new and an old problem is mathematically describable. Herby you can use linguistic fuzzy logic terms (e.g. excellent, good, fair, poor) and online measurement data to specify the attributes of a problem and their specific solution.

These presented related work from computer science with the specification of artificial intelligence and the combination with environmental issues show the potential of this interdisciplinary research topic. The use of knowledge-based systems for diagnosis and prediction of processes enables the process optimization of these with online parameters and experiential knowledge from process operators. Regarding the optimization from the anaerobic digestion process AI methods are used like model-based reasoning and fuzzy logic. The use of case-based reasoning as a knowledge-based decision system with the combination of condition monitoring as real-time control is the focus of this paper.

Methodology

To develop a knowledge-based decision tool to optimize the process from biogas plants following requirements are necessary: understanding of decision making from biogas operators, case study and an artificial intelligence method.

The utilization of experiential knowledge from biogas plant stakeholders to support the decision making, is based on the theory that it is possible to describe a situation with standardized attributes. These attributes characterize situations, technical assessments and organoleptic properties and can be divided in conditions like warm and cold, fast, slow and very slow or pleasant and unpleasant. For the acquisition of experiential knowledge for the optimization of technical processes these methods are used [WEITZE AND KRAFT 2011]:

- interview and questionnaire,
- participant observation,
- experiment and
- inactive monitoring (consideration of documents like laboratory analysis).

With the help of these methods knowledge engineers can acquire important knowledge which is not replaceable with technical measurements. The evaluation of these exemplary methods leads to the description of a theory, to the determination of rules or to acquire of cases. The aim is the conceivable explanation and reproducible description of a case, a rule or a method to transfer this knowledge into a computer-based system. This research topic is described by the human-computer-interaction (HCI) and the artificial intelligence (AI).

The objects of study are a large scale biogas plants and laboratory experiments. For this paper is focused on the input substrates from biogas plants as a parameter which influence the biogas process (Figure 3). Different properties from input materials influence the process but these qualities are not detected with measuring devises. Therefore, the organoleptic examination is a helpful method. The organoleptic examination is a method which use the human senses (eye, nose, tactile sense, ears) to describe the properties of substrates and processes [WEITZE AND KRAFT 2011].

Figure 3 Observation fields (large scale and laboratory experiments, organoleptic examination)

This observation fields enabled the discussion with many stakeholder: biogas plant operators and owner and researcher in the field of biogas plant optimization. With the description of input materials an error diagnosis is the first step for process optimization for the test people. On the other hand the organoleptic examination enables the support and animation for generally perception of the anaerobic process.

With the artificial method case-based reasoning (CBR) a machine language is available. It is possible to classify CBR models in:

- Process diagnosis for the fermentation unit (biological and technical)

- Energy conversion (CHP)
- Input material diagnosis (organoleptic examination)
- Fault detection
- Condition monitoring (technical equipment).

For an easy to learn database a couple of CBR models are necessary which are summarize in a computer-based interface. Various CBR computer software providers are selectable but it is necessary to adopt these models with specific rules and boundaries and attributes. For CBR models plausibility checks are essential. Therefore, some experiments in the laboratory are necessary with some standard physico-chemical analysis and installed measuring devices.

Results and discussion

The biogas operators are well informed about the cause-and-effect-relationship between the characteristics of the input material and the fermentation system. They reduce the new input load if they recognize that the input material is in poor condition. This concerns, for example the water content and the amount of mold. Some operators are also focused on the digestate and evaluate colour and smell as well. This is one result of empirical methods: interview and participant observation.

This paper present the results of the development for a decision making tool for material description with the help of human senses. Analogous to this method cases for process description or fault management is possible. Table 1 shows one structure of an organoleptic based CBR model. It is segmented in object and the description of the object: attribute and range of value from the attribute. For example maize silage is brief presented regarding the object input material.

Table 1 Objects and their descriptions from some knowledge based observations

Object	Description of object	
	Attribute	Range of value
Input material	Name	Maize silage Swine manure Whole grain
Input material e.g. maize silage	Storage	Silo with cover Silo without cover Cesspit
	Colour	Color shade Coloration
	Odour	Intensity Hedonic quality Hedonic intensity
	Morphology	Structure Humidity Particle size Homogeneity
	Source of storage	Top of silo Middle of silo Inside of silo
	[…]	[…]
Output material	Name	Mixed digestate Liquid digestate Dry digestate
Process description	Name	CHP Stirling system Fermenter conditions
[…]	[…]	[…]

For a better understanding of objects and their attributes as a decision tree Figure 4 illustrated the CBR architecture. Concerning the development of a case the guidance or solution for the use of this maize silage is necessary.

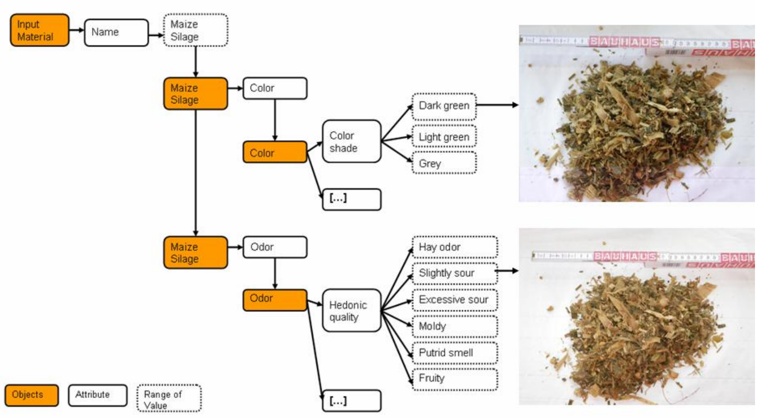

Figure 4 Decision tree for maize silage description (e.g. Colour shade and hedonic quality)

To specify the cases with attributes and solutions interviews and experiments are necessary. All these data were organized in a knowledge-based control system (KBCS) which based on CBR, fuzzy logic and condition monitoring strategies. A prototype of this KBCS is illustrated in Figure 5. It is divided in sources of information, input data, output data and the content of each object. It is based on an object orientated program structure. This architecture show the interaction between the knowledge and the decision making process of the stakeholder, results from experiments and the condition monitoring which use measured data for a process diagnosis.

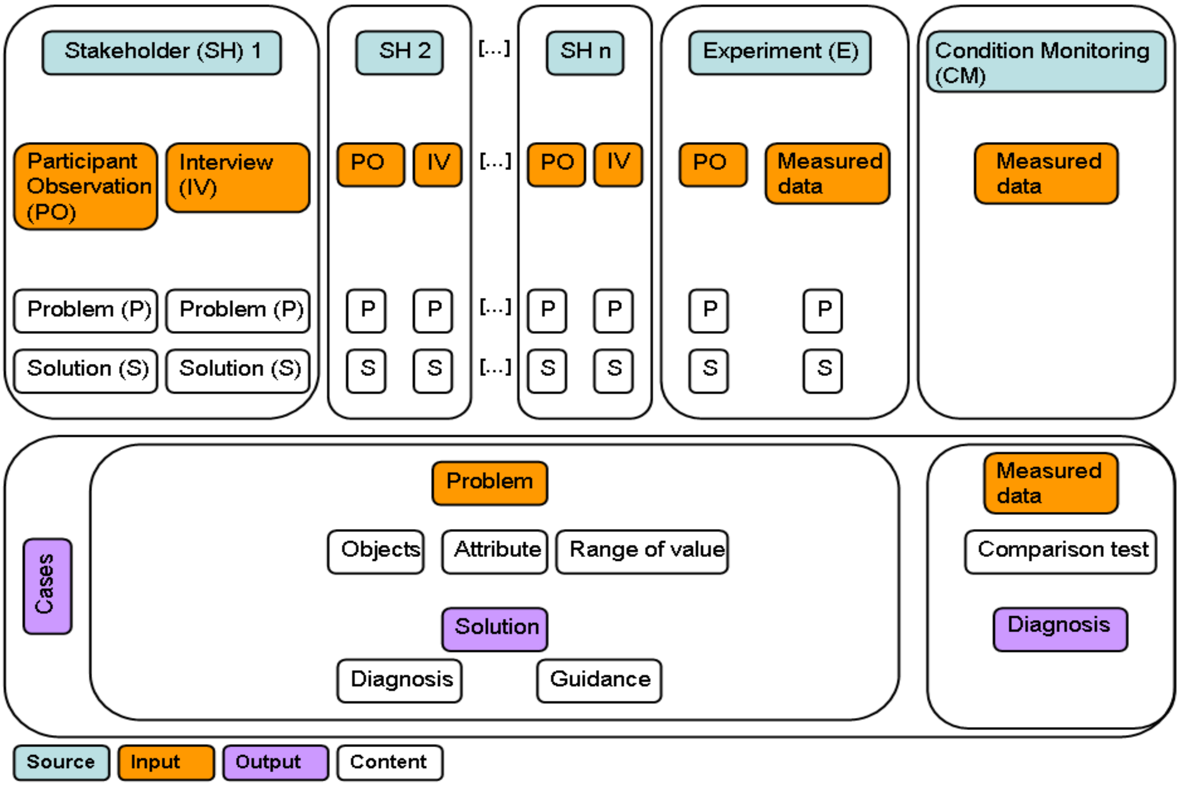

Figure 5 Architecture of the knowledge-based control system for biogas plants

This complex model which based on the artificial intelligence model case-based reasoning enables the integration from experiential knowledge and measured data. It is important to document the decision steps as much as possible in a replicable way. Therefore, the storage of decision processes in cases and the comparison from new situation with the old cases make it possible to develop a big case-based reasoning decision tool. Further research topics are the integration of process monitoring, the condition monitoring and the assessing of output material from biogas plants: digestate. Regarding the condition monitoring the special consideration of technical equipment is necessary to increase the mean time between failures (MTBF).

The method enables the transfer and adaptation for other processes as well if experiential knowledge is involved. Technical questions e.g. operation of composting plants or the integration of a monitoring concept in developing countries for composting processes, landfill operation or waste collection systems are transferable in a knowledge-based control system. During this data collection, empirical methods for knowledge collection and object definition with attribute description are necessary. Case-based reasoning enables the combination of mathematically connections and knowledge-based decision processes in a replicable object orientated model.

CONCLUSIONS

This paper presented the use of artificial intelligence (AI) for process optimization strategies of biogas plants. With a stakeholder analysis internally stakeholder were finding out: biogas plant operators and owner, bank and insurance. Regarding the development of process optimization strategies the decision making of researcher and biogas plant operators are interesting. The presentation of artificial intelligence and deterministic models in environmental issues shows the potential of the utilisation of AI models for process optimization strategies. With the help of the AI method case-based reasoning (CBR) organoleptic examination, experimental results and measuring data in progress are combinable.

The results show the decision tree for maize silage description and the architecture from a knowledge-based control system in general. The use of CBR enables to integrate empirical knowledge in a structured base and figure out a replicable decision making process. This interaction allows the use in other research fields as well. The CBR method is applicable for other technical questions e.g. operation of composting plants or the integration of a monitoring concept in developing countries for composting processes, landfill operation or waste collection systems.

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Assessment of Biogas Generation from Market Food Wastes: Khulna City Perspective

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ABSTRACT

Khulna is the third largest city in Bangladesh with a population of 1000000 within 59.57 square kilometers generating massive quantity of wastes every day. Due to resource and technological limitations, Khulna City Corporation (KCC) has been facing a huge burden to properly manage the whole task of solid waste disposal. Usually, market waste management is not given proper and special attention by the city authority and hence causes enormous problems to the neighboring community adjacent to market places. Biogas offers a sustainable solution to this problem. The aim of this study was to investigate the market wastes disposal practice in Khulna City area and hence proposing a sustainable solution with regards to biogas generation from market food wastes. Major market food wastes in KCC area include: vegetables, fruits, leaves, poultry, slaughter house wastes, etc. This study revealed that around 6000 kg/day of market food wastes are generated in KCC area which accounts for an average energy value of 27.9X10^{6}kJ/day. In the laboratory study, approximately 3690 cm^3 of biogas was obtained from a mixture of 1 kg market food wastes. This study will help in developing new relation between solid waste management and renewable energy from waste.

INTRODUCTION

The management of solid waste represents a major economic and environmental issue throughout the world (Demirbas, 2010). Solid waste problems in developing countries are aggravated by the malfunctioning of traditional waste management systems due to rapid development and the concentration of the population (Deshmukh et al., 2002). As the population and use of resources are higher in urban areas, the rate of waste generation is also high. The root causes of solid waste problems are actually related to the profligate generation of solid waste and the mass production and mass consumption of products without consideration for their final disposal(JICA, 2005). The urban areas of Asia now spend about US$25 billion on solid waste management per year, with this figure increasing to at least US$50 billion in 2025 (World Bank, 1999). There can be no solution to the problems of solid waste until the entire flow of the processes of production, consumption and disposal are considered as an integrated system. There exist an acute relationship between production, consumption, and disposal (JICA,2005). In most of the developing countries this waste problem becomes threatening. The construction of new incineration facilities or landfill sites has become extremely controversial with local opposition and the growing limitations on the total amount of available capacity at current landfill sites(JICA, 2005). These were the additional factors that prompted the search for a more fundamental solution to waste problems (JICA, 2005). Energy from waste is an excellent sollution of this waste problem.

Bangladesh is a small developing country with many problems. At present the waste management becomes a major issue to solve. The waste disposal practice is very poor and environmet pollution is occured largely due to improper waste disposal. Biogas generation from waste is an sustainable way to fight aganst waste problem. Municipal solid waste consume around 68% to 81% of food and vegetables (Alamgir et al., 2007). By anaerobic digestion process can convert food waste into biogas. Biogas typically refers to a gas produced by the biological breakdown of organic matter in the absence of oxygen. Organic waste such as dead plant and animal material, animal feces, and kitchen waste can be converted into a gaseousfuel called biogas. Biogas originates from biogenic material and is a type of bio fuel (Wikipedia). Biogas is about 20 % lighter than air and has an ignition temperature in the range of 650^{0}C to 750^{0}C. It is odorless and colorless gas that burns with clear blue flame similar to that of LPG gas. Its caloric value is 20 Mega Joules (MJ) /m^3 and burns with 60 % efficiency in a conventional biogas stove. The gas is a mixture of methane (CH_4) 65 %, carbon dioxide 25 %; hydrogen 5 %, nitrogen 3 %, hydrogen sulphide (trace), water vapor 0.3 % (Figure 1).The gas is useful as a fuel substitute for firewood, dung, agricultural residues, petrol, diesel, and electricity, depending on the nature of the task, and local supply conditions and constraints.

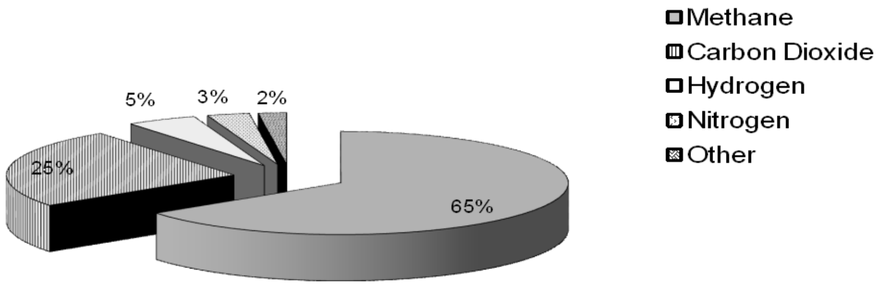

Figure 1 Composition of Biogas (in%)

Biogas system also provide a residue organic waste, after anaerobic digestion that has superior nutrient qualities over the usual organic fertilizer, cattle dung, as it is in the form of ammonia.Conventional biogas system is practiced mainly using cow dung, but there is a great scope to utilise food waste inbiogas generation. Market, locally called kacha Bazar is a big source of food waste and there waste can be collected continuously. It is expected from this research that the scenarios in khulna city and assessment of biogas will help to develope a realization of importance of biogas from organc food wastes of bazar area.

METHODOLOGY

Selection of Study Area

Khulna, the third largest city of Bangladesh, is located in the southern part of the country and is situated below the tropic of cancer, around the intersection of latitude 22.49°N and longitude 89.34°E (Figure 2). The area of Khulna city is 47 square km with a population 1.5 million (BBS, 2009). With regards to investigating wastes problem in bazar a field survey was conducted in the Khulna city area.

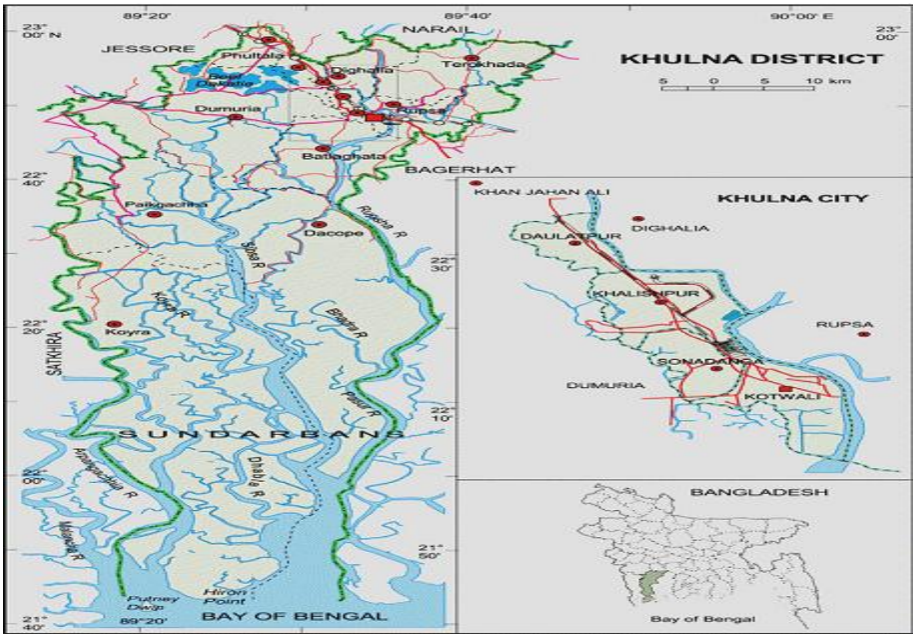

Figure 2 Location map of Khulna City

In khulna city, there are more than 12 kacha bazar (locally called) in different places.Gollamari bazar, Moilapota bazar, Rupsha kacha bazar, Mistripara bazar, New market bazar, boyre bazar, boikali bazar, chitrali bazar, khalishpur bazar, Dawlutpur bazar, phulbarigate bazar, shiromoni bazar. For the field survey 8 bazar ware selected and these are gollamari bazar, moilapota bazar, rupsha bazar , new market bazar, mistripara bazar, khalishpur bazar, dawlotpur bazar and phulbarigate bazar.

Field Survey in the Selected Area

In this study, a field investigation was carried out in Khulna city area to investigate how amount organic wastes produce in bazar. During this investigation, a questionnaire form was drafted along the lines of another study (Moniruzzaman, 2007). A thorough classification of the shops was performed on the basis of food items like vegetables, poultry, and meat. The amount of waste produced eevery day in bazar, information about where and how this waste dispose were obtained from the questionnaire survey. Finally, the gross average quantities of different wastes were determined in kg d^{-1}.

Laboratory Experiment

In the laboratory, test was performed to estimate how amount biogas can generate from bazar waste. for this test anaerobic digestion process was applied and the used equipments were,

- Container(2.75 L) as reactor
- Cylindrical standed plastic jar(100 ml)
- Plastic bottle
- Rubber cork
- Pipe(6.25 ml)
- Color water

A containers of 2.75 L were selected for the reactor. The wastes were collected locally and cut into smaller pieces (approximate size 0.25 in). The mixture was then fed into the reactor (glass cylinder) through the inlet. The inlet of the reactor was made airtight by using rubber cork for anaerobic digestion. A small hole was cut through the rubber cork. The diameter of the hole was 6.25mm. A small dia (6.25mm) pipe was inserted through the hole cut in the rubber cork.(Figure 3) A plastic bottle and a cylindrical standard plastic jar (1000ml) was used for the gas measurement set up. The plastic bottle was first filled with colored water. The color was prepared by mixing color with the clear water. The cylindrical plastic jar was also filled with the colored water to the top of this jar. The cylindrical jar was set invertly with full water to the plastic bottle. The reason of the colored water use was that it will help to mark the water level from the transparent plastic jar. The point of the plastic pipe (6.25mm) was set to the upper point of the inverted cylindrical plastic jar. When the gas is generated it flows through the pipe from the reactor to the cylindrical jar. The accumulated gas in the cylindrical jar forces the water level to lower from the initial height. The total lowering amount (ml) from the initial height of the jar represents the amount of generated gas. This is the procedure of gas measurement.

Figure 3 Experimental set up for gas production

RESULTS AND DISCUSSION

From questionnaire survey and observation of disposal system in the selcted market a clear concept about present waste disposal systen in all bazar and how amount waste produce in every day was obtained. In most bazar the seller collects lots of vagetables in the bazar everyday and at the end of the day not quiet little amount they throw away as rotten. Table 1 shows the approximate amount of waste in bazar in khulna city from questionier survey. Due to resource and technological limitations, Khulna City Corporation (KCC) has been facing a huge burden to properly manage the whole task of solid waste disposal and hence affect the public health and hygienic. Most of the time wastes are dumped on nearest open area or by the road side.The gollamari bazar is near moyourakhhi river and there is usual practice to dispose all the bazar waste to this river. For this practice of waste disposal now this river is fully polluted and the river water is out of use. The sorrounding environment is inhygenic. In some bazar the wastes are dumped on the road side which is complete misuse of road and road users suffer a lot when they pass by this road (Figure 4). Somewhere it also shown that waste from poultry and meat are directly dumped on drain. In rainy season open dumping of waste cause a seriousn problem. All the drainage system blogged due to wastes. City corporation collect wastes from some bazar and finally disposed by landfill at Rajbandth. In below the real picture of present bazar situation is show

Table 1 Wastes in study areas

Bazar	Waste(kg)			Energy Value (kJ)
	Vegetables	Poultry	Meat	
Gollamari Bazar	180	84	400	3.08x10^6
Newmarket Bazar	114	80	300	2.29x10^6
Moilapota Bazar	160	64	150	1.74x10^6
Rupsha Bazar	300	50	200	2.26x10^6
Mistripara Bazar	100	60	120	1.30x10^6
Khalishpur Bazar	200	84	250	2.48x10^6
Dawlutpur Bazar	250	40	325	2.86x10^6
Phulbarigate Bazar	50	120	150	1.49x10^6

Fig 4 Waste disposal practice in bazar

The effect of improper disposal is serious on environment. On the other hand disposal to landfills results in the production of methane gas which is 21 times as potent a greenhouse gas as carbon dioxide. The main disadvantages of improper diposal of wastes is inhygenic and polluted environment. From questionier survey the selsmen in bazar are not so much intersted in disposing the waste properly. Every morning when they start their business in bazar they clean their places and dumped nearest open area or in the bins. In most bazar there is a bazar development committee but they have no awareness about this situation. Figure 5 shows the flow diagram of existing market food wastes disposal practice in Khulna city and the advarse effect on the environment and soorroundings.

After field ovservation adopting continuous process in the laboratory experiment for biogas from bazar food wastes was done. From the laboratory experiment, it is found that a huge amount biogas can be produced from bazar wastes. Table 2 shows the experimental result of biogas which prouced from bazar wastes.

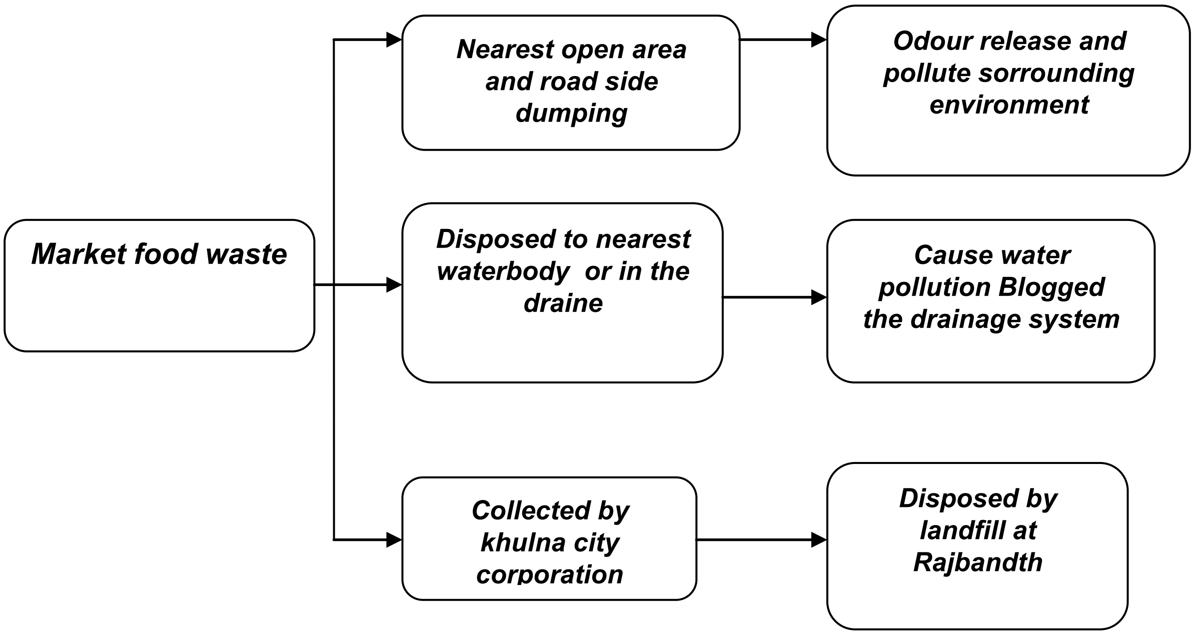

Figure 5 Flow diagram showing the existing market food wastes disposal practice in Khulna city

Table 2 Bio-gas generation in the laboratory study

Time (days)	Gas generated (ml)	Cumulative Gas (ml)	Time (days)	Gas generated (ml)	Cumulative Gas (ml)
1	345	345	7	953	5227
2	817	1162	8	1505	6732
3	1395	2557	9	802	7534
4	746	3303	10	697	8231
5	387	3690	11	421	8652
6	584	4274	12	314	8966

The graph (Figure 6) represents the pattern of gas generation from bazar wastes. Initial weight of sample was 1kg and after new feeding at day 6the total weight was 2kg. From the graph it is obseved that at begining the gas production rate increase with time but after certain time when the digestion precess become slow, the gas production rate also decrease. At the day 6 new feeding was added to the reactor and the gas production again increases. After 5 days the total amount of produced gas was 3690 ml.

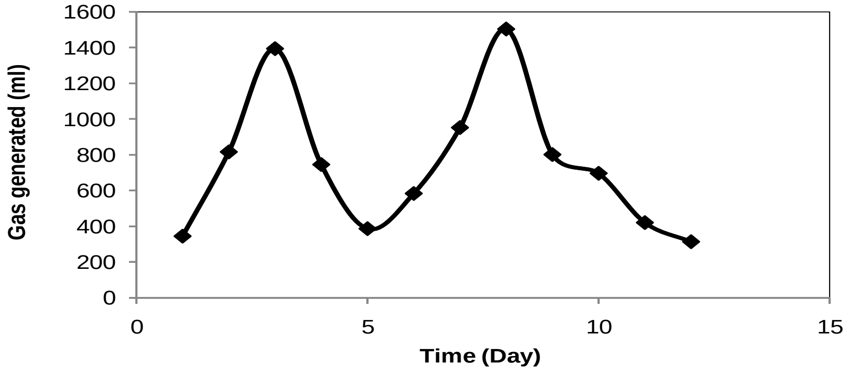

Figure 6 Generation of bio-gas from market food wastes

Figure 7 describes the total cumulative bio-gas production in 12 days. The total amont after 12 days was 8966 ml. From this graph it is clear that the if continuous process can adopt then a continuous gas supply can be obtained.

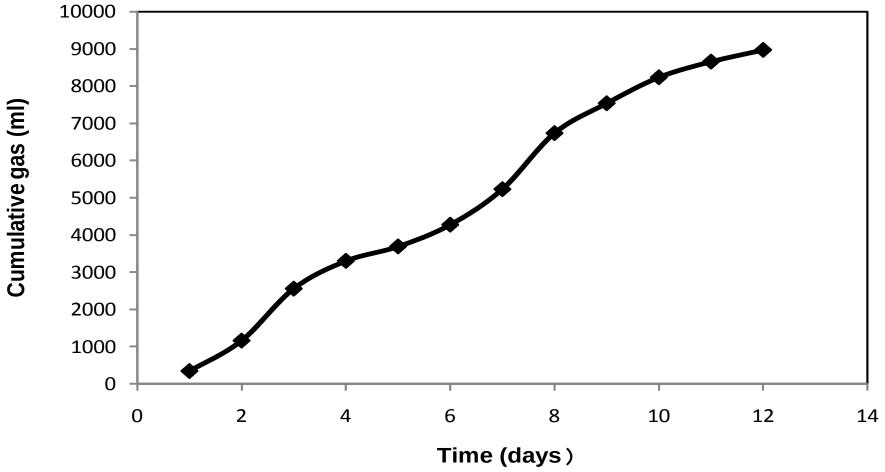

Figure 7 Cumulative generation of bio-gas from market food wastes

CONCLUSIONS

This study represented the scenario of existing market food wastes disposal practice in Khulna city area which was largly responsible in polluting sorrounding environment. Biogas generation from market food waste was deemed a sustainable way to fight aganst environmental burdens. An estimated *6000* kg/day of market food wastes were generated in KCC area which accounted for an average energy value of *27.9*X10^{6}kJ/day. Laboratory batch experiment suggested that approximately 3690 cm^3 of biogas was could be obtained from a mixture of 1 kg market food wastes. High energy value of food wastes generated biogas would help in reducing the dependence on fossil fuel. Locally it is quite possible to make biogas plant near any market place and the produced biogas can be used in fuel purpose or in making electrycity. This study revealed that adoption of natural anaerobic process for bio-gas production would be lowcost. Furthermore, a continuous process for bio-gas generation would have promizing application with regards to renewable energy conservation to face the challenges of environmental sustainability in the 21st century.

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Geo-environmental Aspects of Liner in Municipal Solid Waste Landfills

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ABSTRACT

Landfills in recent times are highly engineered containment systems, designed to minimize the impacts of solid waste trash or garbage, refuse or rubbish on the environment and human health. In modern landfills, the waste is contained by a liner system. The functions of liners and the drainage layer are complementary and work together to prevent the uncontrolled discharge of contaminants to the environment. The main goal of this study is to implement the liner system to isolate the landfill contents and therefore, to protect the soil and groundwater from pollution. Rajbandh landfill as a sanitary landfill serves the remedy for potential threat caused by conventional damping. Rajbandh landfill composed with a double composite clay liners upon natural soil assemble with PVC leachate collection pipe into leachate treatment tank surrounding by drainage material and air vent pipe for removal of toxic air; makes it environment friendly. By the conceptual design proposals, the findings of the geo-environmental assessment and environmental monitoring results, risk assessment and the conclusions reached in investigations. The fundamental objective behind waste management is that of sustainability. It is implicit therefore that landfill development and operation (which are intrinsically linked) should reflect this approach.

INTRODUCTION

Landfills are sites that permanently store waste in a manner that minimizes the release of contaminants into the environment. The environmental containment systems for municipal solid waste landfills include a system beneath the waste and a cover system constructed over the waste. The system beneath the waste generally consists of a combination of a drainage layer overlaying low-permeability barriers or liners. The functions of liners and the drainage layer are complementary and work together to prevent the uncontrolled discharge of contaminants to the environment. The drainage layer or leachate collection and removal system is an integral part of the environmental protection system of any liner. Leachate is the contaminated liquid in a landfill that passes through or has been in direct contact with solid waste. The leachate collection system is placed directly below the waste for the purpose of collecting the liquid above the liner. If this contaminated liquid is not removed from the landfill it creates a hydraulic head or downward force on the liner that drives the liquid out of the landfill. The leachate collection system collects and conveys leachate within the landfill to controlled collection points or sumps to limit the buildup of leachate or hydraulic head pipes to convey the leachate to the leachate removal system. The leachate removal system may contain networks of sumps, pumps, flow meters, and other flow conveyance and monitoring components for the purpose of removing the leachate on top of the liner. The purpose of the liner is to impede leachate and gas migration out of a municipal solid waste landfill and improve the collection capability of the overlying drainage layer. Liners installed beneath modern municipal solid waste landfills generally consist of a recompacted earth liner, a single-composite liner or double-composite liner. The leak detection layer is a layer between the two liners. The purpose of the leak detection layer is monitoring the performance of the upper liner and allows appropriate action to be taken when leachate is found in this layer. This liner system has a leachate collection system directly above the primary liner.

(a) (b)

Figure 1: Rajbandh (a) Conventional (b) Sanitary landfill

OBJECTIVE

The study covers the following objectives:

- The primary objective of landfill site design is to provide effective control measures to prevent or reduce as far as possible negative effects on the environment, in particular the pollution of surface water, groundwater, soil and air, as well as the resulting risks to human health arising from landfilling of waste.
- Assessment of ground conditions, the geology and hydrogeology of the site, the potential environmental impacts and the location of the landfill.
- To compare the landfill operation between conventional landfill to sanitary landfill after use of liner.

MATERIALS & METHOD

Leachate control within a landfill involves the following steps:

(a) Prevention of migration of leachate from landfill sides and landfill base to the subsoil by a suitable liner system; and

(b) Drainage of leachate collected at the base of a landfill to the sides of the landfill and removal of the leachate from within the landfill.

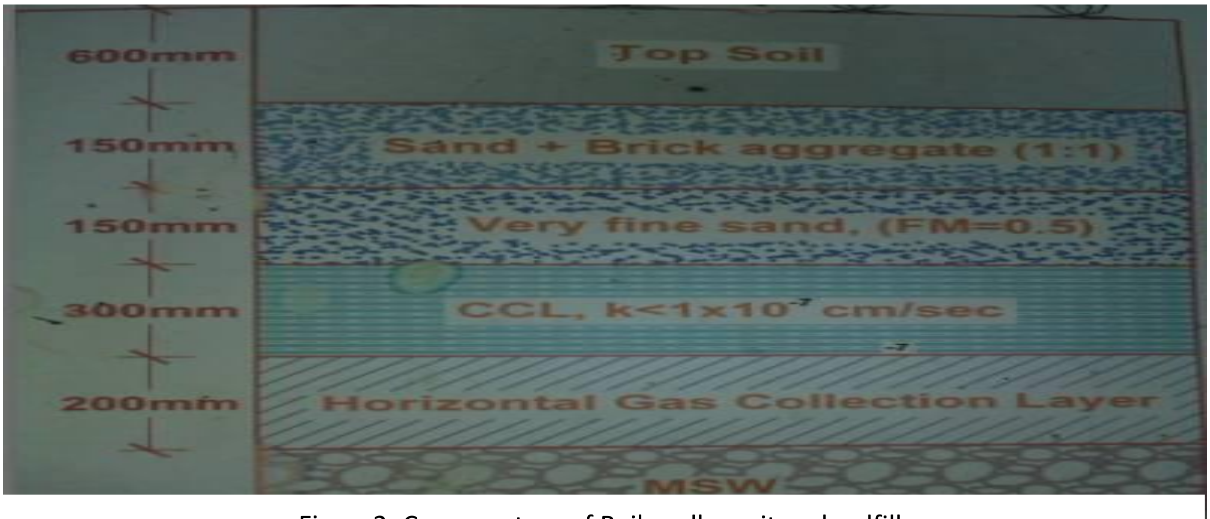

Figure2: Cover system of Rajbandh sanitary landfill.

Liner systems comprise of a combination of leachate drainage, collection layers and barrier layers (Fig. 2). A competent liner system should have low permeability, should be robust and durable and should be resistant to chemical attack, puncture and rupture. A liner system may comprise of a combination of barrier materials such as natural clays, amended soils and flexible geomembranes.

STUDY AREA

Rajbandh landfill is one of the largest damping site located at about 7km away from Khulna city by the side of Khulna-Shatkira highway. It is a very good place for damping municipal waste for outside Khulna city. It's a final disposal site and carrying major solid wastes of total city. Trucks and other vehicles are supplied by Khulna metropolitan city authority, collects wastes from different waste bin in city area.

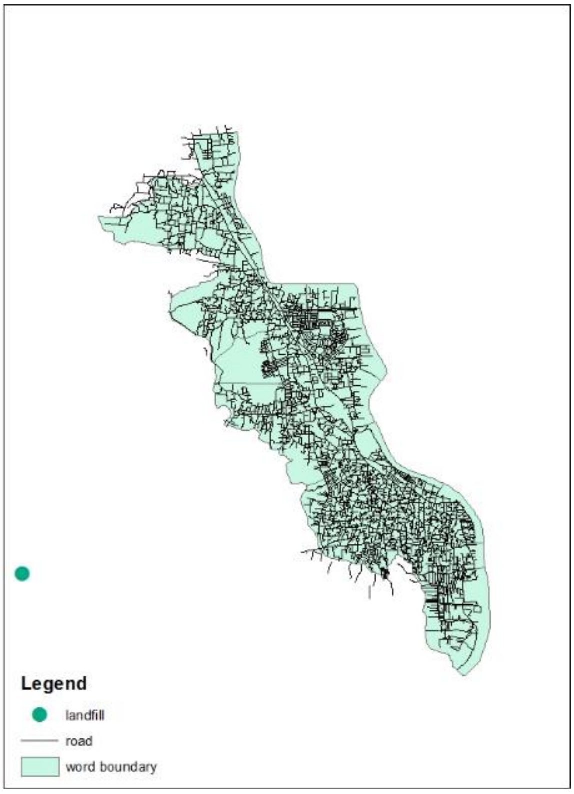

Figure3 : Rajbandh landfill site located in latitude 22.49^{0}North and longitude 89.34^{0}East by Arc GIS

ESSENTIAL COMPONENTS

The seven essential components of a MSW landfill are:

(a) A liner system at the base and sides of the landfill which prevents migration of leachate or gas to the surrounding soil.

(b) A leachate collection and control facility which collects and extracts leachate from within and from the base of the landfill and then treats the leachate.

(c) A gas collection and control facility (optional for small landfills) which collects and extracts gas

1. Diameter of PVC leachate collection pipe will be 100mm.
2. Drainage material of surrounding leachate collection pipe will be washed brick aggregate 20-32mm dia.
3. Top half of leachate collection pipe is perforated.
4. Compaction of CCL will performed in two layers, 200mm X 200mm

From within and from the top of the landfill and then treats it or uses it for energy recovery.

(d) A final cover system at the top of the landfill which enhances surface drainage, prevents infiltrating water and supports surface vegetation.
(e) A surface water drainage system which collects and removes all surface runoff from the landfill site.
(f) An environmental monitoring system which periodically collects and analyses air, surface water, soil-gas and ground water samples around the landfill site.
(g) A closure and post-closure plan which lists the steps that must be taken to close and secure a landfill site once the filling operation has been completed and the activities for long-term monitoring, operation and maintenance of the completed landfill.

ENVIRONMENTAL IMPACT AND ITS MINIMIZATION

The impact of dumping municipal solid waste on land without any containment such dumps cause the following problems:

(a) Groundwater contamination through leachate.
(b) Surface water contamination through runoff.
(c) Air contamination due to gases, litter, dust, bad odor.
(d) Other problems due to rodents, pests, fire, slope failure, erosion etc.

Landfills minimize the harmful impact of solid waste on the environment by the following mechanisms:

(a) Isolation of waste through containment;
(b) Elimination of polluting pathways;
(c) Controlled collection and treatment of products of physical, chemical and biological changes within a waste dump both liquids and gases; and
(d) Environmental monitoring till the waste becomes stable.

(a) (b)

Figure5: (a) leachate collection and treatment tank,(b) Air vent pipe for removal of toxic air

RESULT AND DISCUSSION

The objective of an environmental monitoring system is (a) to find out whether a landfill is performing as designed; and (b) to ensure that the landfill is conforming to the regulatory environmental standards. Monitoring at a landfill site is carried out in four zones: (a) on and within the landfill; (b) in the unsaturated subsurface zone beneath and around the landfill; (c) in the groundwater (saturated) zone beneath and around the landfill and (d) in the atmosphere/local air above and around the landfill. The parameters to be monitored regularly are:

(i) Leachate head within the landfill;
(ii) Leachate and gas quality within the landfill;
(iii) Long-term movements of the landfill cover;
(iv) Quality of pore fluid and pore gas in the vadose zone;
(v) Quality of groundwater in the saturated zones and
(vi) Air quality above the landfill, at the gas control facilities, at buildings on or near the landfill and along any preferential migration paths.

The indicators of leachate quality and landfill gas quality must be decided after conducting a study relating to the type of the waste, the age of the waste, the composition of leachate and gas likely to be generated and the geotechnical as well as hydro-geological features of the area.

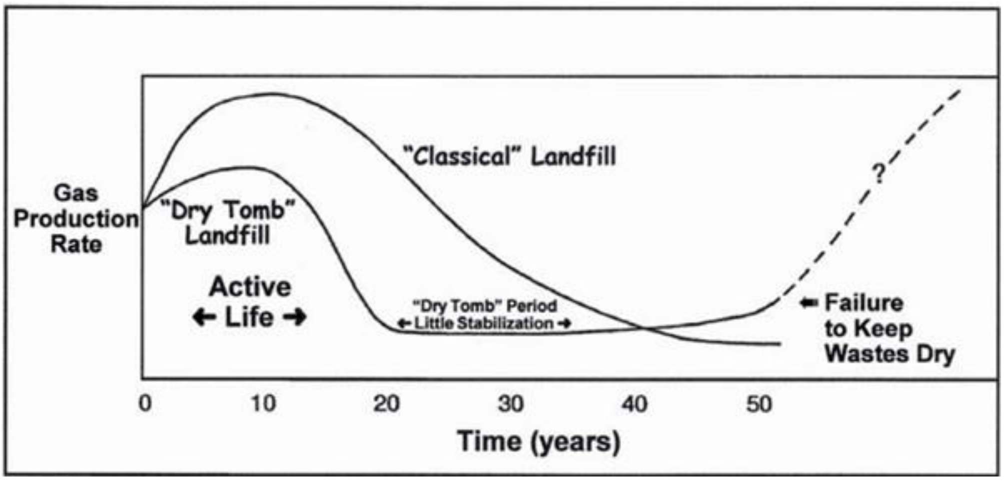

Figure 6: Comparison of Pattern of Landfill Gas Generation over Time at Classical Sanitary and "Dry Tomb" Landfill (Lee et.al. 1991)

The frequency of monitoring will vary from site to site but it must be so fixed that it is capable of detecting unusual events and risks in the initial phases of their appearance so as to give time to diagnose and localize the cause and enable early steps to be taken for containment or remediation. Usually a monthly or a bi-monthly monitoring frequency is considered suitable during the operational phase of a landfill as well as for 3 to 4 years after closure; this frequency can be decreased to 2-3 times a year in later years, if all systems perform satisfactorily. The monitoring frequency may have to be increased if higher concentrations than expected are detected, if control systems are changed or if drainage systems become clogged/non-functional. The frequency of monitoring may also be increased during those periods in which gas generation or leachate generation is higher, such as during the monsoon periods.

IMPACT OF A LANDFILL ON AREA HYDROLOGY

The construction of a large landfill will significantly impact the groundwater recharge in the area covered by the landfill. This altered recharge could significantly alter the groundwater hydrology in the region. It could change the direction of groundwater flow by changing watershed divides and the water table which could alter the velocity of groundwater movement. This, in turn, could have a significant adverse effect on adjacent properties utilizing groundwater as a water supply as well as on adjacent areas where groundwater becomes an important part of the surface water hydrology such as in wetlands. Some consulting firms attempt to mediate this impact by constructing groundwater recharge areas in which surface water running off the cap from the closed landfill is deliberately recharged to the groundwater at the edge of the landfill. Such systems will require perpetual maintenance to assure that the landfill does not adversely affect the groundwater flow regimes in the region. Of particular concern is the potential for plugging of the recharged galleries by eroded materials in the surface run off.

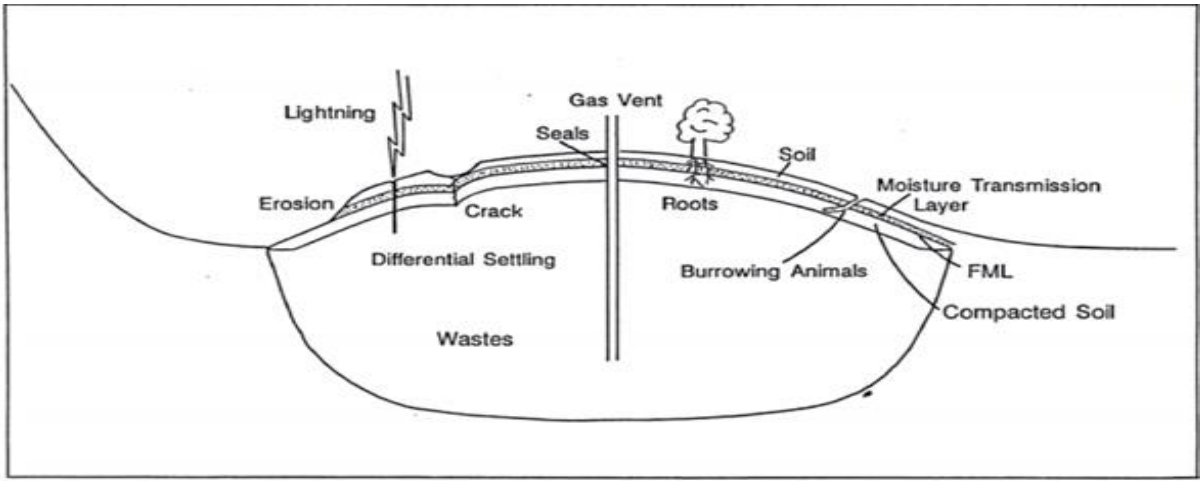

Figure 7: Factors Affecting Long-Term Integrity of Landfill Cap.

It is important to emphasize that if a landfill is located near a wetlands area, that significant damage to the wetlands systems could occur by the landfill altering the groundwater flow regimes that discharge in the wetlands. It is inappropriate, as sometimes advocated by landfill proponents, to claim that since the recharge system provides the same amount of annual average recharge to the groundwater as occurred prior to the construction of the landfill, that the construction of the landfill will not have an adverse effect on the wetlands. Wetlands ecosystems are sensitive to seasonal flow regimes. Therefore, great caution must be exercised in constructing landfills in areas where the landfill could significantly impact the amounts of groundwater delivered to a wetlands at any time during the year. It will be virtually impossible to construct landfills in such areas without having an impact on the wetlands system.

REMEDIAL MEASURES FOR THE GROUNDWATER POLLUTION

From the groundwater monitoring it is clearly evident that the leachate generated from the landfill site is affecting the groundwater quality in the adjacent areas through percolation in the subsoil. Therefore, some remedial measures are required to prevent further contamination. This can be achieved by the management of the leachate generated within the landfill. Leachate management can be achieved through effective control of leachate generation, its treatment and subsequent recycling throughout the waste. Engineered landfill sites are generally provided with impermeable liner and drainage system at the base of the landfill, which will not allow leachate to percolate into subsoil. All the leachate accumulated at the base of the landfill can be collected for recycling or treatment. This collected leachate can be distributed throughout the waste by means of spraying the leachate across the landfill surface. Some of the water may be lost through evaporation and therefore leading to reduction in the volume of the leachate for ultimate treatment. Techno economic feasibility studies should be carried out for choosing the options for a landfill site. In non-engineered landfill, is neither having any bottom liner nor any leachate collection and treatment system. Therefore, all the leachate generated finds its paths into the surrounding environment. In such conditions only feasible options that could be followed are:
(i) Limiting the infiltration of the water through the landfill cover by providing impermeable clay cover. Due to this less water will enter and subsequently less leachate will be generated, thereby reducing the amount of leachate reaching the landfill base.

(ii) Extraction of the leachate collected at the base can be done and it can be recycled, so that less amount will enter the aquifer lying below.
(iii) Increasing the evapo-transpiration rate by providing vegetation cover over the landfill can also reduce leachate production.

DISCUSSION AND CONCLUSIONS

There is an urgent need for additional solid waste management capacity because the current design, operation, closure, and post closure activities for sanitary landfills does not provide for adequate public health and environmental protection of the groundwater resources near the landfill. First, if the regulatory agencies wish to persist with trying to keep the solid wastes dry, then the landfill must be lined, capped, and maintained for as long as the wastes represent a potential for groundwater contamination in such a manner as to prevent such contamination. It is suggested that municipalities in areas where there is limited space for new sanitary landfills should adopt recycling of wastes to the maximum extent readily achievable (about 50%) and then mass burn incineration of the remaining solid wastes. This approach should include the most readily available, highest efficiency air pollution control on the incinerator air emissions. The ash from these incinerators should be fixed with cement based reagents and placed in monofils which are properly lined, capped, operated, and maintained during operation and in post closure to assure that at no time in the future will the contaminants in the ash lead to groundwater contamination. First and foremost is the need to ensure that true long-term public health and environmental protection will be achieved through the proposed approach for solid waste management. This is typically not being done today. Hopefully, regulations will soon be developed and implemented which would achieve this objective.

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Assessment of Parameters Involved in Leachate Pollution Index and Evaluation of Contaminant Potential of Landfill Lysimeter Using Delphi Technique

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ABSTRACT

Leachate pollution index (LPI) is a quantitative tool by which the leachate pollution data of solid waste landfill can be evaluated uniformly. LPI can be also used as a tool to assess the leachate pollution potential from landfill particularly at places where there is a high risk of leachate migration and pollution of groundwater. The main focus of this research is to estimate of an index known as LPI as well as to evaluate the contamination potential using LPI of a landfill lysimeter at KUET campus, Khulna, Bangladesh. The parameters required to calculate LPI were discussed in terms of their variations over time and the significance level. The LPI of three lysimeter tests set-up representing both the open dump and sanitary landfill conditions have been calculated. Result reveals that among the four lysimeter operating systems, collection system of open dump lysimeter had the highest LPI, while detection system had the lowest LPI. Moreover, LPI had been decreased in relation to the increase of elapsed period of MSW deposited in landfill lysimeter similar to pilot scale sanitary landfill constructed at Khulna and opening dumping practices. In contrary, LPI for this landfill lysimeter has been compared with the LPI estimated for treated leachate standards. Finally, it can be concluded that LPI for landfill lysimeter was significantly high and proper treatment is required before discharging into the natural streams..

Keywords*:* Landfill lysimeter, open dump, sanitary landfill, contaminant potential, leachate parameter, leachate pollution index, treated leachate.

INTRODUCTION

Landfills are the primary means of municipal solid waste (MSW) disposal in many countries worldwide because they offer dumping high quantities of MSW at economical costs in omparison to other disposal methods such as incineration (Rafizul et al. 2012). The term ‘landfill’ is used herein to describe a unit operation for final disposal of MSW on land, designed and constructed with the objective of minimum impact to the environment. This term encompasses the other terms such as ‘secured landfill’ and ‘engineered landfills’ which are also sometimes applied to MSW disposal units (Tubtimthai 2003). The term ‘landfill’ can be treated as synonymous to ‘sanitary landfill’ of MSW, only if the latter is designed on the principle of waste containment and is characterized by the presence of a liner and leachate collection system to prevent ground water contamination. Sanitary landfill is one of the secure and safe facilities for the disposal of MSW; however, it needs high standard of environment protection in the operation of landfill (Davis and Cornwell 1998). The main aim of a sanitary landfill is to use it for a longer time for disposal of solid waste with less negative effect to the ecosystem (Rafizul et al. 2012). Moreover, it is a well-suited method for managing of MSW all over the world and to investigate the performance of sanitary landfill the behavioral patterns namely; leachate generation, landfill gas (LFG) emissions etc. are required (Visvanathan et al. 2002).

Landfill leachate produced form MSW landfill sites is generally heavily contaminated and consist of complex wastewater that is very difficult to deal with (Daud et al. 2009). If the landfill is uncontrolled condition there exists threat to get mixed of leachate with groundwater. Threats to groundwater from the unlined and uncontrolled landfills exist all over the world, particularly in the underdeveloped and developing countries where hazardous industrial waste is also co-disposed with MSW and no provision of separate landfills for hazardous waste exist (Alamgir et al. 2005; Rafizul et al. 2011). The most commonly reported danger to the human health from the MSW landfills is from the use of groundwater that has been contaminated by leachate (Chian and DeWalle 1976; Kelley 1976; Lo 1996; Kumar et al. 2002; Kumar and Alappat 2005c). Even if there are no hazardous wastes placed in solid waste landfills, the leachate is still reported as a significant threat to the groundwater (Lee 2002). This threat is severe when the landfill site is located near rivers and streams which are a major source of agriculture and productivity, industrial and domestic water supply. The effect of landfills on the environment due to inappropriate landfill operations and maintenance are as follows: a) the uncontrolled emission of LFG that cause greenhouse effect on the atmospheric and odor problems in the neighborhood; b) uncontrolled effluent from solid waste landfill know as leachate might migrate and pollute the ground or surface water sources; and c) the breeding and harboring of disease vector in improperly managed landfills (Tchobanoglous et al. 1993). The generation of leachate is a result of percolation of precipitation through open landfill or through cap of the completed site (Aziz et al. 2007). Moreover, the characteristics of leachate are highly variable depending on the wastes deposited in the landfill, composition of wastes, moisture content, the particle size, the degree of compaction, sampling procedures, landfill design and operation, the hydrology of the site, the climate, and age of the fill and other site-specific conditions including landfill design and type of liners used, if any (Rafizul et al. 2011; Leckie et al. 1979). Organic content of leachate pollution is generally measured in terms of biological oxygen demand (BOD5) and chemical oxygen demand (COD). The concentrations of leachate contaminants may range over several orders of magnitude (Deng and Englehardt 2006). A combination of pollutants (BOD5, COD, ammonia, inorganic salts, etc.) in higher concentrations renders landfill leachate as a potential source of contamination both to ground and surface waters, hence necessitates its treatment prior to discharge to water resources (Bashir et al. 2010).

The management of leachate is among the most important factors to be considered in planning, designing, operation, and long term management of an MSW landfill (Halim et al. 2010). Leachate can contaminate groundwater where landfills are not provided with liners and surface water if it is not collected and treated prior to its discharge. The overall pollution potential of MSW landfill leachate can be calculated in terms of leachate pollution index (LPI) as proposed by Kumar and Alappat (2005a). Because identification and quantification of pollutants in landfill leachate is the major limitation for its successful treatment (Trankler et al. 2005). LPI can be used as a mean to determine whether a landfill requires immediate attention in terms of introducing remediation measures. Most of the landfills in developing countries including Bangladesh are not designed with proper leachate collection mechanism. Moreover, most of the landfills do not come under sanitary landfill classification because there are no facilities for collection and/or treatment of leachate and there is no infrastructure to collect landfill gas. The state regulatory authorities in almost of the countries have framed regulations to safe guard against the contamination of groundwater sources from the leachate generated from the landfills (Umar et al. 2010). As the remedial and preventive measures are so expensive for most of the countries of the world, so it is to be taken up in a phased manner. Thus a system needs to develop to classify the landfill in the basis of their hazard potential. That is how the concept of leachate pollution index has come. Kumar and Alappat (2003b) have developed a technique to find the leachate contamination potential of different landfills on a comparative scale in terms of LPI. LPI has many potential applications including ranking of landfill sites, resource allocation, trend analysis, enforcement of standards, scientific research and public information (Sharma et al. 2008). The formulation process and complete description on the development of the LPI has been discussed elsewhere (Kumar and Alappat 2003a). The LPI represents the level of leachate contamination potential of a given landfill. The present study was carried out to determine and compare the LPI for leachate collected from pilot scale open dump and sanitary landfill lysimeter as well as the derived LPI for other researches for the similar cases available in the literature. Moreover for easy to estimate and compare the leachate contamination potential of different solid waste landfill either it is open or sanitary condition in a given geographical area would be a useful tool in this regard.

LANDFILL LYSIMETER EXPERIMENT

The landfill lysimeter test facilities were set-up in the geo-environmental research station at the backyard of Civil Engineering Building, KUET, Khulna, Bangladesh. In this study three different situations of landfill were considered as well as both the open dump lysimeter-A having a base liner and sanitary landfill lysimeter-B and C at two different types of cap liner were simulated. The lysimeter-A represents the present practice of open dumping of MSW as well as lysimeter-B and C as the sanitary landfill facilities in Bangladesh. Moreover, the function of provided cap liner in sanitary lysimeter-B and C were (i) to promote the accelerate of surface runoff; (ii) eliminating percolation of water through cap liner; (ii) to eliminate the production of leachate and (iv) helps for vegetation growth that minimize the erosion and promote transpiration of water back to the atmospheres and landfill gas control, while the function of base liner is to protect the infiltration of leachate in to the underlying layer.

Refer to Figure 1, as the technical details of reference cell for the construction of desired three lysimeters. In reference cell, the deposited MSW mainly consists of 93 (w/w) organic (food and vegetables), 3 (w/w) of plastic/polythene and 2 (w/w) of leather/rubber, 1 (w/w) of animal bone and rubber/leather as well as 1 (w/w) of rope/straw and egg pill. The organic content and moisture content of the deposited MSW were found 52 % and 65 %, respectively, and the total volume was 2.80 m3 (height 1.6 m) with a manual compaction to achieve the unit weight of 1,064 kg/m3. At the bottom of reference cell, a concrete layer of 125 mm thick was provided then the lysimeters were filled with stone chips (diameter 5-20 mm) and coarse sand (diameter 0.05-0.40 mm) to the height of 15 cm of each to ensure uniform and uninterrupted drainage. The clay used as CCL having the percentages of constituents of sand, silt and clay of 10, 56.6 and 33.4 %, respectively, while the value of optimum moisture content and maximum dry density of 18 % and 16 kN/m3, respectively, and the coefficient of hydraulic conductivity of 1.90x10-7 cm/sec (Rafizul et al. 2009).

Lysimeter was used to simulate the different landfill concept, operational condition and the total weight of MSW was deposited in each lysimeter as presented in Table 1. The three landfill lysimeter , were constructed using the brick wall of 250 mm thick having outer and inner diameter of approximately 1.98 m and 1.48 m, respectively, with a total height of 3.35 m, resting on a 250 mm thick of reinforced cement concrete mat foundation at a depth of 760 mm below the existing ground surface. The landfill lysimeter was plastered both the inner and outer sides with two coatings of waterproofing agent to avoid leakages and corrosion due to acidic environment. Further, the sanitary landfill lysimeters (B and C) consists of landfill gas (LFG) collection system above the MSW and leachate recirculation system below the MSW in landfill lysimeter. At the bottom of each landfill lysimeter, a cement layer of 125 mm thick was provided then the landfill lysimeters were filled with stone chips (diameter 5-20 mm) and coarse sand (diameter 0.05-0.40 mm) to the height of 15 cm each to ensure proper leachate drainage. At the base of each landfill lysimeter after placing the perforated leachate collection pipe, a geo-textile sheet having 0.60 m wide and 1.65 m length was placed to avoid rapid clogging by the sediments from landfill lysimeter. A leachate collection tank (3.68 x1.56 x1.64 m) accommodating four separate leachate discharge pipes in the temporary collection and storage containers, were constructed using 250 mm thick brick wall. Brief descriptions of the constructed landfill lysimeter having three different conditions were given in followings.

Table 1 Operational condition of lysimeter to simulate different landfill conditions

lysimeter	Operating condition	Refuse (kg)	Liner specification	Simulation
A	Open dump lysimeter with leachate detection (A1) system Open dump lysimeter with leachate collection (A2) system	2860	400mm thick CCL as a barrier between leachate detection and collection system of lysimeter-A	present practice of open dumping
B	Sanitary lysimeter with gas collection and leachate recirculation system	2985	Cap liner-I (300mm thick CCL)	applicability of designed top cover
C		2800	Cap liner-II (900mm thick natural top soil)	

Landfill Lysimeter-A (Open Dump)

The type and volume of MSW deposited in open dump lysimeter-A was exactly the same as deposited in the reference cell. In open dump lysimeter-A, a CCL of 400 mm thick was placed as the base liner and a layer of compost of 150 mm thick was used as the top cover to simulate the behavior of present practice of open dumping in Bangladesh as depicted in Figure 2. In this lysimeter the MSW was not covered by a top cover system to pervert the movement of air, water and generated LFG. Moreover, the thickness of the deposited MSW in lysimeter-A is such that it is expected the atmospheric air can move in the entire MSW deposited in this cell with negligible inference. Due to the mentioned practical situations, lysimeter-A, represents the open dump condition comparing the counterparts i.e sanitary alndfill lysimeter-B and lysimeter-C.

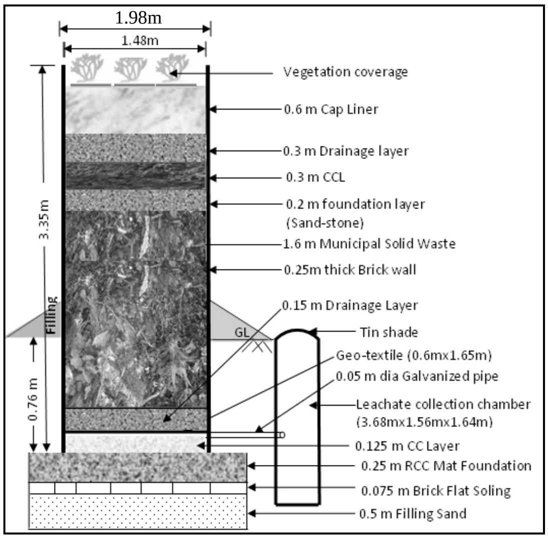

Figure 1 Schematic diagram of reference cell for landfill lysimeter design

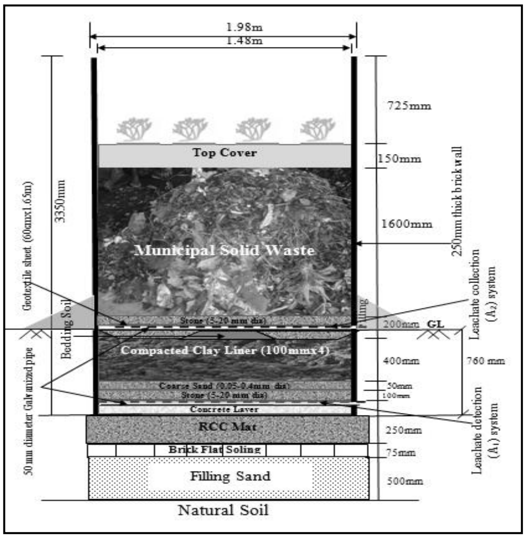

Figure 2 Schematic diagram of open dump lysimeter-A

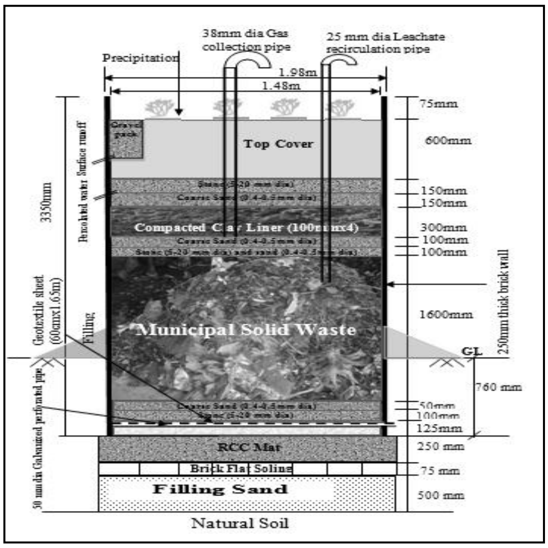

Figure 3 Schematic diagram of sanitary landfill lysimeter-B

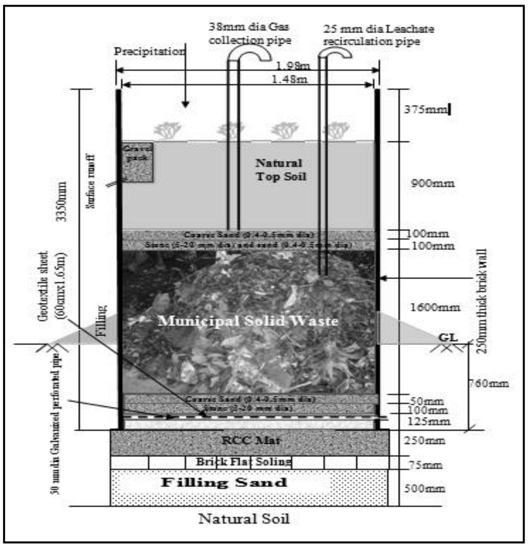

Figure 4 Schematic diagram of sanitary landfill lysimeter-C

Sanitary Landfill Lysimeter-B (Cap liner I)

In sanitary landfill lysimeter-B as detailed in Figure 3, the characteristics and volume of the deposited MSW was similar to that of the open dump lysimeter-A. However, it differs with open dump lysimeter-A, by a top cover and without having a base liner, because this cell aims to examine the applicability of the designed top cover. The top cover consists of stone chips (diameter 5-20 mm) and coarse sand (diameter 0.05-0.40 mm) layer each of 100 mm thickness, then a 300 mm thick CCL was provided. On the CCL, there were 150 mm thick coarse sand (diameter 0.05-0.40 mm) and 150 mm thick stone chips (diameter 5-20 mm), which was followed by 600 mm thick top soil. In sanitary landfill lysimeter-B, 38 mm diameter of gas collection and 25 mm diameter of leachate recirculation pipe were installed. During the installation of these pipes and penetration through the top cover, special arrangements i.e. disc shaped rubber gasket having 3 mm thickness and 300 mm diameter was used for the protection of any possible leakage between the pipe and surrounding soil media.

Sanitary Landfill Lysimeter-C (Cap liner II)

In sanitary landfill lysimeter-C, there was also no base liner and the provided top cover was different than that of the sanitary landfill lysimeter-B. In this case no CCL was used; however, 900 mm thick natural top soil was used instead of 300 mm CCL and 600 mm thick top soil as depicted in Figure 4. Moreover, the drainage and gas collection layers were remained same as the sanitary landfill lysimeter-B. In sanitary landfill lysimeter-C, 38 mm diameter of LFG collection and 25 mm diameter of leachate recirculation pipe was set-up, similar to that of sanitary landfill lysimeter-B. Designated compaction of the CCL in the lysimeter means the degree of compaction which was provided in the pilot scale sanitary landfill (PSSL). To achieve the designated compaction at the CCL of lysimeter, locally manufactured hammer similar to that used in the PSSL was employed.

LABORATORY INVESTIGATIONS AND METHODOLOGY

Leachate samples were collected at a regular interval of times from leachate collection chamber accomplished of four distinct collecting system such as detection (A1) and collection (A2) system of open dump lysimeter-A as well as collection system of sanitary landfill lysimeter-B and C. In the laboratory, pH was determined by pH meter (HACH, Model No. Sens ion 156), chloride by potentiometric titration method using silver nitrate solution, BOD5 by BOD meter (HACH, HQ-40d), TCB by filter membrane system, Arsenic using sulfamic acid and zinc powder as well as COD by closed reflexive method as per the standard method (APHA 1998). In addition, total dissolved solid (TDS) dried at 103-105oC, ammonia nitrogen (NH4-N) by nesselerization standard method and total kjeldahl nitrogen (TKN) by macro-kjeldahl method as per the standard method (APHA 1998) were determined in the laboratory. Moreover, Fe, Cu, Zn, Cr, Ni and Pb were analysed using spectrophotometer (HACH; DR/2400) in accordance with the standard method (APHA 1998).

In contrast, a study conducted by Kumar and Alappat (2004) and advocated that for calculating LPI, the weighted sum linear aggregation function was found to be the most suitable. When the data for all the pollutant variables included in LPI is not available (m < 18), the LPI can be calculated using data set of the available pollutants by the Equation (1). The averaged subindex curves for all the pollutant variables have been reported by Kumar and Alappat (2003a) for evaluating sub index score of the leachate pollutant variable.

$$LPI = \frac{\sum_{i=1}^{m} w_i p_i}{\sum_{i=1}^{m} w_i} \qquad (1)$$

Where, LPI is the weighted additive leachate pollution index, m is the leachate pollutant parameter for which data is available, wi is the weight for ith pollutant variable and pi is the sub index score of the ith leachate pollutant variable. Where pollutant parameter for which data is available in this study as, m < 18 (15) and $\sum_{i=1}^{m} w_i < 1$ (0.824) (Table 3, Column 2).

Table 2 Characteristics of leachate generated from MSW in landfill lysimeter

Parameter	Operating condition of landfill lysimeter			
	A1	A2	B	C
pH	5.98-8.17 (6.77)	6.14-8.02 (7.25)	6.40-8.61 (7.42)	6.30-8.38 (7.48)
Chloride	970-3097 (2238)	1155-3572 (2395)	510-1870 (1086)	110-1810 (1181)
TDS	210-9930 (5720)	140-35810 (12507)	1800-29980(9786)	420-26000 (13132)
TKN	195-1120 (594.52)	251-1430 (785.61)	480-2187 (1307.07)	312-1860 (1014.31)
NH4-N	187-705 (428.50)	127.9-901.3 (547.57)	190-997 (653.26)	187-965.7 (598.34)
BOD5	83-6750 (2310.43)	143-20420 (5176.5)	437-12358 (3308.7)	490-16733 (4428.3)
COD	160-20800 (2498)	800-60000 (8425)	320-60000 (5654)	300-60000 (11137)
TCB	98-6540 (2468.70)	145-8398 (3101.33)	152-8200 (3065.52)	85-8230 (2895.89)
Fe	1.8-70 (19.5)	3.4-82.6 (24.2)	3.6-91 (27.9)	1.4-72 (6.3)
As	0.01-0.02 (0.01)	0.01-0.05 (0.03)	0.01-0.03 (0.01)	0.01-0.02 (0.01)
Zn	0.25-1.267 (0.76)	0.15-0.97 (0.54)	0.10-0.55 (0.31)	0.10-0.576 (0.2)
Cu	0.04-0.98 (0.44)	0.04-0.97 (0.34)	0.04-0.76 (0.20)	0.04-0.60 (0.18)
Cr	0.01-0.07 (0.05)	0.01-0.056 (0.04)	0.023-0.03 (0.06)	0.01-0.025 (0.04)
Pb	0.12-0.32 (0.20)	0.179-0.397 (0.3)	0.25-0.78 (0.44)	0.10-0.476 (0.3)
Ni	0.04-0.075 (0.05)	0.04-0.055 (0.05)	0.045-0.09 (0.07)	0.04-0.07 (0.05)

Note: TDS=total dissolve solid, TKN=total kjeldahl nitrogen, NH4-N=ammonia nitrogen, BOD5= biological oxygen demand, COD=chemical oxygen demand, TCB= total colifom bacteria, Fe=iron, As=arsenic, Zn=zinc, Cu=copper, Pb=lead and Ni=nickel. All values in mg/L, except pH and TCB ((cfu/100ml)). Range is given for the minimum and maximum values, while the value of parenthesis indicates the mean values.

RESULTS AND DISCUSSIONS

Significance and Assessment of Leachate Parameters Involved in LPI

The leachate parameters involved in LPI for the detection (A1) and collection (A2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C are presented in Table 2. The ranges were given for the minimum and maximum values, while the value of parenthesis indicates the mean values. Moreover, the significance, assessment and comparison of the evaluated leachate parameters involved in LPI were analysed and discussed in followings.

PH

The pH in leachate is considered to be the most significant parameter affecting leachate concentration in MSW landfill (Bilgili et al. 2007). Qasim and Chiang (1994) suggested that increase of pH in leachate with time as a result of the decrease in the concentration of volatile fatty acids (VFA). Moreover, the increase in pH is due to onset of methanogenic activity as a result increase in methane gas production and decrease in hydrogen, carbon dioxide and VFA (Kim 2005). Murphy et al. (1995) reported that pH in leachate <7.0, was in the acidic side of pH scale, but after in the scale of >7.0, waste leachate had alkaline pH. Moreover, a study conducted by Christensen et al. (2001) and observed that leachate is generally found to have pH between 4.5 and 9.. Initial low pH is due to high concentration of VFAs (Bohdziewicz et al. 2008). Stabilized leachate shows fairly constant pH with little variations and it may range between 7.5 and 9. Moreover, Kulikowska and Klimiuk (2008) and Tatsi et al. (2002) reported similar range of pH from old landfill sites, that is, 7.46–8.61 and 7.3–8.8, respectively. In contrary, the values of pH in leachate from lysimeter at varying operational condition were provided in Table 2. Result reveals that pH was ranging from 6.33 to 7.75 and 7.38 to 8.61 for the collection system (A2) of open dump lysimeter-A and sanitary landfill lysimeter-B, respectively, in the first 100 day of entire lysimeter operation. After 100 day, pH began to increase and it was reached to 8 for the collection system (A2) of open dump lysimeter-A. A study conducted by Abbas et al. (2009) and advocated that pH of young leachate is less than 6.5, while old landfill leachate has pH higher than 7.5. Moreover, consumption of volatile fatty acids (VFA's) in MSW, produced during the acid anaerobic phase, increases pH in between 7 and 8 Farquhar (1989). So, the range and increase of pH for the collection system of open dump lysimeter-A were agreed well with the postulation stated by Abbas et al. (2009) as well as Farquhar (1989). After that, no considerable changes were observed of pH in leachate and it was measured between 6 and 8. These results were also in accordance with the data stated by Cossu et al. (2003); Ishigaki et al. (2003) and Nakasaki et al. (1993). However, to depict the validity of the findings in this study then also compared with those results in case of open lysimeter (OL) and closed lysimeter (CL) postulated by Trankler et al. (2005) and Visvanathan et al. (2007). The values of pH for the collection system of sanitary landfill lysimeter-B was higher than that of the collection system of open dump lysimeter-A and it was continued as well as similar trend was also observed for the same cases of lysimeter studied by Trankler et al. (2005) and Visvanathan et al. (2007).

Nitrogen

The TKN is the sum of organic nitrogen (Organic-N) and NH4-N (APHA 1998). The NH4-N is representative of inorganic matter (Rafizul et al. 2012). The landfill lysimeter leachate contains low concentration of organic nitrogen and very high concentration of NH4-N (60-90 % of TKN (Johansen et al. 1976; Lema et al. 1988). Moreover, according to Kulikowska and Klimiuk (2008) ammonium represents the major proportion of total nitrogen. Moreover, NH4-N, the great majority of TKN (Lema et al. 1988), did not decreasing trend in concentration with time. The ammonia concentration released from waste by the decomposition of proteins and there is no mechanism for its degradation under Methanogenenic condition. So, only leaching can decrease the NH4-N concentration (Kjeldsen et al. 2002).

In contrary, the concentration of TKN and NH4-N in leachate from lysimeter at varying operational condition were provided in Table 2. Result reveals that both the concentration of TKN and NH4-N in leachate of lysimeter at varying operational condition was decreased in relation to the increase of elapsed period of MSW deposition in lysimeter. A study conducted by Abbas et al. (2009) and reportred that in comparison to soluble organics, the release of soluble nitrogen from waste into leachate continues over longer period. As a result the concentration of NH4-N decreases with the increase in age of the landfill which is due to hydrolysis and fermentation of nitrogenous fractions of biodegradable refuses substances. So, the findings are agreed well with the postulation given by Abbas et al. (2009). According to Speece (1996) and Jokela et al. (2002) such a decrease of NH4-N could be attributed to the ammonium bicarbonate alkalinity, which in turn maintains a pH close to neutral in the anaerobic solid waste bioreactors. Moreover, the higher values of TKN and NH4-N were recorded for the collection system of sanitary landfill lysimeter-B against the other counterparts i.e. sanitary landfill lysimeter-C and open dump lysimeter-A (Table 2). The open dump lysimeter-A produces low TKN concentrations by reason of dilution of pollutant due to its top cover design enhance rainfall percolation and similar trend was also observed by Tubtimthai (2003). These findings were also suppoted by Van Velsen (1979) and he has reported that acclimation to high concentration of NH4-N was possible in sanitary (anaerobic) reactors. However, the maximum concentrations of TKN (1120, 1430, 2187 and 1860 mg/L) and NH4-N (705, 901.3, 997 and 965.7 mg/L) were recorded within the 90th day of lysimeter operation of detection and collection system of open dump lysimeter-A as well as the collection system of sanitary lysimeter-B and C, respectively. Chu et al. (1994) too have reported that the NH4-N of landfill leachate ranged between 500 and 1500mg/L after a period of 3-8 years and will remain at this level at least for the next 50 years.

BOD5 and COD

Jones et al. (2006) reported that in the initial acidogenic biodegradation stage, the leachate is characterized by high BOD5 and COD. Young landfill leachate is characterized by high BOD5 (4000–13,000 mg/L) and COD (30,000–60,000 mg/L) (Foo et al. 2009). According to Tatsi et al. (2002), young leachate may have BOD5 as high as 81,000 mg/L. A much higher value of COD (70,900 mg/L) is reported in leachate obtained from the Thessaloniki Greater Area (Greece) by Tatsi et al. (2002). A decrease in BOD5 and COD is often reported with the increase in age of the landfill. For stabilized leachates, COD generally ranges between 5000–20,000 mg/L (Li et al. 2002).

The BOD5/COD ratio provides a good estimate of the state of the leachate and this ration for young leachate is generally between 0.4-0.5 (Kurniawan et al. 2006). During the methanogenic phase, the organic strength of the leachate is reduced by methanogenic bacteria such as methanogenic archaea and the concentration of VFAs also declines which results in a ratio of BOD5/COD less than 0.1 (Kurniawan et al. 2006 and Rivas et al . 2004).

In contrary, the concentration of BOD5 and COD in leachate from lysimeter at varying operational condition were provided in Table 2. The concentrations of BOD5 and COD were initially about 20000 mg/L and 60000 mg/L, respectively, for the collection system of open dump lysimeter-A and it were dropped with the increase of elapsed period. Similay trend were also observed for other operational condidion of landfill lysimeter-B (BOD5(12000 mg/L) and COD (60000 mg/L)) and C (BOD5 (17000 mg/L)and COD (60000mg/L)) (Table 2). This high readings indicate that there are organic materials in the open dump lysimeter-A which are highly biodegradable (Mizanur et al., 1999). It was further supported by Chian and DeWalle (1976) and advocated that due to the initially biodegradable nature, organic compound decreases more rapidly than that of inorganic compound with increasing the age of landfill. A young landfill generates a leachate which has high BOD5 and COD, but this strength decreases rapidly after a short period of time (i.e. around one year). For example, the concentration of BOD5 and COD in the leachate at the age of 6 and 9 months is 10,000-20,000 mg/L and 20,000-40,000 mg/L, respectively while, the concentration of BOD5 and COD in the leachate at the age of one and 1.5 years is 1,560-1,800 mg/L and 5370-7040 mg/L, respectively (Agamuthu, 2001). Also, the pollution level of leachate decreases as the age of leachate increases (Curi et al., 1999). Moreover, the variation in leachate quality with age should be expected throughout most of the landfill life because organic matter will continue to undergo stabilization (Qasim and Chiang 1994). A decrease of COD occurs over the elapsed period of landfill and it can be attributed to a combination of reduction in organic contaminants and the increased biodegradation of organic compounds (Krug and Ham 1995). A constant decrease in COD is also expected as degradation of organic matter continues (Ehrig 1989).

Chlorides

Chloride in leachate is considered to be a very conservative anion that is only attenuated during dilution. Ehrig (1983) suggested that there is no observable difference of chloride between acidogenic and methanogenic phases of solid waste landfill. According to Deng and Englehardt (2007), the concentration of chlorides may range between 200 to 3000 mg/L for a 1-2 year old landfill and the concentration decreases to 100 to 400 for a landfill greater than 5–10 years old. The values of chloride in leachate from lysimeter at varying operational condition were provided in Table 2. Result reveals that collection system of lysimeter-A had the highest concentration of chloride than that of other lysimeter operational condition. Chu et al. (1994) reported that chloride concentration increases with the increasing age of waste fill. Its release is not controlled by factors such as pH (Komilis et al., 1999). It is interesting to note that pH and Cl- concentrations began to increase simultaneously in all the cases of sanitary landfill lysimeter operation. This situation can be explained by the findings of Manning and Robinson (1999). As a result of the increase in pH, the dissolution of chloride increases and thus the chloride concentration in the leachate increases. The values of pH and Cl- concentrations increase at the same operation days of 100, and the findings are agreed well with the statement stated by Manning and Robinson (1999). Bowman et al. (2002) found chlorides as high as 8000 mg/L in Newington landfill leachate in Sydney. Because chlorides are usually not attenuated by soil and are extremely mobile under all conditions, they have a special significance as the tracer element of leachate plume linking the groundwater (Kumar and Alappat 2005b).

TDS

TDS comprises mainly of inorganic salts and dissolved organics. TDS is one of the parameters taken into consideration for licensing discharge of landfill leachate in many countries such as the Koshy et al. (2008). The amount of TDS reflects the extent of mineralization and a higher TDS concentration can change the physical and chemical characteristics of the receiving water. The increase in salinity due to increase in TDS concentration also increases toxicity by changing the ionic composition of water. The values of TDS in leachate from lysimeter at varying operational condition were provided in Table 2. Based on observed results it can be depicted that the highest values of TDS of 9930, 33801, 29980 and 26000 mg/L were recorded for the detection and collection system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C and then it was dropped with the increasing of elapsed period. A study conducted by Yuen (2001) and reported that TS concentration is expected to decrease as the leachate moves from acidogenic to methanogenic but the TDS concentrations do not change in large quantities as total solid. So, the decreasing treand of TDS in this study were aggreed well with the statement given by Yuen (2001). The solids joining the leachate at the later stage are the products of chemical and biological degradation of waste .The concentration of TDS in leachate depends not only on elapsed period from the waste deposition but also the types, applicability and compactness of base and cap liner (Mollekopf et al. 2002 and Diaz et al.1996).

Heavy Metals

The leachates from MSW landfills are highly contaminated and have higher concentration of organic and inorganic pollutants as well as toxic substances of metals (Kettunen et al. 1998). The main processes for the low metal concentrations in leachate are sorption and precipitation. The solubility of metals with sulfides and carbonates is low and sulfide precipitation is often cited as an explanation for low concentrations of heavy metal in leachate (Bozkurt et al. 1999). In general, the concentration of heavy metals in landfill leachate is fairly low (Christensen et al. 2001). The concentration of heavy metal viz Fe, As, Zn, Cu, Cr, Pb and Ni in leachate from lysimeter at varying operational condition were provided in Table 2. Result reveals that initially the concentration of evaluated heavy metal was high, but, it was started to decrease with the increasing of elapsed period. Concentration of heavy metals in landfill lysimeter is generally higher at earlier stages because of higher metal solubility as a result of low pH caused by production of organic acids Kulikowska and Klimiuk (2008). As a result of decreased pH at later stages, a decrease in metal solubility occurs resulting in rapid decrease in concentration of heavy metals except lead because lead is known to produce very heavy complex with humic acids (Harmsen 1983). A lower concentration of Ni (0.13mg/L), Zn (0.61 mg/L), Cu (0.07mg/L), Pb (0.07 mg/L), and Cr (0.08 mg/L) were found in 106 Danish landfills by Christensen et al. (2001) and Kjeldsen and Christophersen (2001). The high dissolved organic carbon (DOC) concentration in leachate renders the metals nontoxic because only the free metals are known to exhibit toxicity (Vigneault et al. 2005 and Koukal et al. 2003). However, the solubility and mobility of metals may increase in the presence of natural and synthetic complexing ligands such as EDTA and humic substances (Jones et al. 2006). Further, colloids have great affinity for heavy metals and a significant but highly variable fraction of heavy metals is associated with colloidal matter (Christensen et al. 2001; Jensen and Christensen 1999).

According to Baun and Christensen (2004), less than 30%, typically less than 10% of the total metal concentration is present in free metal ion forms and the rest is present in colloidal or organic complexes. Jensen and Christensen (1999) found that 10–60% of Ni, 30–100% Cu and 0–95% Zn were constituted in colloidal fractions.

Interpretation of Parameters involved in Individual and Overall LPI

The leachate characteristics as well as the individual and overall LPI obtained for a particular elapsed period of 7 day from MSW deposited in landfill lysimeter at different operational condition of leachate detection (A1) and collection (A2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C is provided in Table 3. Since the data for all the parameters required for LPI was not accessible, the LPI has been calculated on the basis of the available data. Table 3 depicts that concentration of Cr, BOD5, As, Zn, TKN, Ni, TCB, TDS, Cu, chloride and Fe was higher for the collection system of lysimeter-A, while the collection system of sanitary landfill lysimeter-B contains higher Pb, pH and NH4-N. A significant difference between the individual and overall pollution ratings for both the collection system of open dump lysimeter-A and sanitary landfill lysimeter-B was observed due to the distinct difference to their leachate concentrations. Here, it can be noted that concentration of leachate parameters involved in LPI for the detection (A1) and collection (A2) system of open dump lysimeter-A, differs and it implies the significance difference of individual and cumulative pollution ratings between these two lysimeter operating system. Moreover, the concentrations of As, Ni and Cu were fairly similar for both the collection system of sanitary landfill lysimeter-B and C. Although these two lysimeter exhibited notable differences for Cr, BOD5, Zn, TDS, chloride and Fe, but the influence of individual and cumulative polluting rating is insignificant.

The concentrations of Cr, COD, BOD5, As, Zinc, TKN, Ni, TDS, Cu and chloride in leachate for the collection system of open dump lysimeter-A were higher than that of the collection system of sanitary landfill lysimeter-C which resulted in significantly higher cumulative pollution rating. However, comparatively the higher Cr, BOD5, As, Zn, TKN, Ni, TCB, TDS, Cu, chloride and Fe significantly influenced the cumulative pollution ratings of collection (A2) system of open dump lysimeter-A. Result reveals that difference of Cr, As, Zn, pH, Ni, Cu and Fe concentration was found to be significant but, their individual pollutant rating was insignificant for entire lysimeter operating system. As the individual pollution ratings of COD, TKN, NH4-N, TDS and chloride were found to be lower for the detection (A1) system of open dump lysimeter-A, the cumulative pollution rating of lysimeter-A was consequently lower. Based on Table 3, it can also be concluded that Cr, COD, BOD5, TKN, TDS and Fe for the collection (A2) system of sanitary landfill lysimeter-A was more than 2.76, 2.65, 1.38, 2.16, 3.61 and 1.76 times higher than detection (A1) system of lysimeter-A, and consequently the cumulative pollution rating for collection system was 1.35 times higher than detection system of lysimeter-A. Here, it can be noted that all the concentration in leachate except Zn, Cu and chloride in case of the detection (A1) system of open dump lysimeter-A were found to be the lowest than that of the collection system of lysimeter-A, B and C and had the lowest individual and cumulative pollution rating as well as consequently the lower LPI (Table 3).

In contrary, the parameters involved in LPI, individual, cumulative pollutant rating and consequently the overall LPI of the entire lysimeter operating system for the elapsed period 120 and 900 days are provided in Tables 4 and 5, respectively. Table 4 reveals that concentration of leachate in case of the collection (A2) system of open dump lysimeter-A was found to be the highest and it implies the highest overall LPI than that of other operating system. Moreover, all the concentrations in leachate except Cr, Zn, Ni, NH4-N, Cu and Fe in case of the detection (A1) system of open dump lysimeter-A has the lowest and it implies the lower individual, cumulative and overall LPI than that of other lysimeter operating system. Here, it is important to note that concentration of Cr, Pb, Zn, pH, Cu and Fe differs significantly with each of lysimeter operating system, however, the difference of their individual pollutant rating was negligible. Moreover, the concentration of Fe is varied widely to each other, but, their individual pollutant rating difference was fairly insignificant (Table 4). In contrast, the Table 5 reveals that concentration of leachate, individual and cumulative pollutant rating as well as the overall LPI was fairly same for the operating condition of landfill lysimeter. Here, it is important to note that concentration of pollutant parameter in leachate decreases with the increase of days after filling of MSW in lysimeter. Moreover, Table 5 depicts that leachate concentration at elapsed period 900 days was comparatively lower and which implies the lower LPI than the previous elapsed period (Table 3 and 4).

Table 3 Calculation of LPI at 7 day after filling of MSW in landfill lysimeter

Pollutant variable	Variable weight, wi	Pollutant concentration, ci				Individual pollutant rating, pi				Overall pollutant rating, wipi			
		A1	A2	B	C	A1	A2	B	C	A1	A2	B	C
Chromium	0.064	0.076	0.21	0.17	0.09	5	5	5	5	0.32	0.32	0.32	0.32
Lead	0.063	0.41	0.55	0.92	0.65	7	7	9	8	0.441	0.441	0.567	0.504
COD	0.062	22650	60000	60000	56490	84	94	94	93	5.208	5.828	5.828	5.766
Mercury	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
BOD5	0.061	2080	2860	2790	2286	41	46	45	42	2.501	2.806	2.745	2.562
Arsenic	0.061	0.01	0.04	0.03	0.02	5	5	5	5	0.305	0.305	0.305	0.305
Cyanide	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
Phenol	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
Zinc	0.056	1.4	1.5	0.98	0.65	5.5	5	5	5	0.308	0.28	0.28	0.28
pH	0.055	6.87	7.87	7.92	7.38	6	5	5	6	0.33	0.275	0.275	0.33
TKN	0.053	1010	2180	1430	1340	33	78	50	45	1.749	4.134	2.65	2.385
Nickel	0.052	0.1	0.19	0.13	0.12	5	5	5	5	0.26	0.26	0.26	0.26
TCB	0.052	6540	8280	8200	8230	85	89	90	90	4.42	4.628	4.68	4.68
NH4-N	0.051	705	897	997	920	77	95	99	97	3.927	4.845	5.049	4.947
TDS	0.05	9876	35670	29120	26580	21	83	69	63	1.05	4.15	3.45	3.15
Copper	0.05	1.3	1.6	1.1	1.2	7	7	7	7	0.35	0.35	0.35	0.35
Chlorides	0.049	3037	3572	1350	760	23	31	12	8	1.127	1.519	0.588	0.392
Total Iron	0.045	25.9	45.7	43.5	38.7	5.5	6	6	6	0.2475	0.27	0.27	0.27
Summation	0.824									22.54	30.41	27.62	26.50
Derived LPI										27.36	36.91	33.52	32.16

Note: COD=chemical oxygen demand, BOD5= biological oxygen demand, TKN=total kjeldahl nitrogen, NH4-N=ammonia nitrogen, TCB= total colifom bacteria and TDS=total dissolve solid. All concentrations are in mg/L, except pH and total coliform bacteria unit (cfu/100ml) and NA= Leachate concentration not available.

Table 4 Calculation of LPI at 120 day after filling of MSW in landfill lysimeter

Pollutant variable	Variable weight, wi	Pollutant concentration, ci				Individual pollutant rating, pi				Overall pollutant rating, wipi			
		A1	A2	B	C	A1	A2	B	C	A1	A2	B	C
Chromium	0.064	0.03	0.02	0.04	0.04	5.00	5.00	5.00	5.00	0.32	0.32	0.32	0.32
Lead	0.063	0.20	0.35	0.58	0.42	6.00	7.00	7.50	7.00	0.38	0.44	0.47	0.44
COD	0.062	160	6760	1440	5120	8.50	68.00	44.00	67.00	0.53	4.22	2.73	4.15
Mercury	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
BOD5	0.061	83	3575	980	2179	8.00	51.00	27.00	41.00	0.49	3.11	1.65	2.50
Arsenic	0.061	0.02	0.04	0.02	0.02	5.00	5.00	5.00	5.00	0.31	0.31	0.31	0.31
Cyanide	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
Phenol	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
Zinc	0.056	1.20	0.97	0.12	0.11	5.00	5.00	5.00	5.00	0.28	0.28	0.28	0.28
pH	0.055	6.75	6.42	7.84	6.30	6.00	7.00	5.00	7.00	0.33	0.39	0.28	0.39
TKN	0.053	945	1637	1556	1218	27.00	58.00	54.00	40.00	1.43	3.07	2.86	2.12
Nickel	0.052	0.07	0.04	0.06	0.05	5.50	5.50	5.50	5.50	0.29	0.29	0.29	0.29
TCB	0.052	6212	7945	7898	7460	85.00	87.00	88.00	86.00	4.42	4.52	4.58	4.47
NH4-N	0.051	610	654	787	698	66.00	73.00	88.00	85.00	3.37	3.72	4.49	4.34
TDS	0.05	4440	13440	5810	8300	9.00	34.00	13.00	18.00	0.45	1.70	0.65	0.90
Copper	0.05	0.12	0.08	0.05	0.08	5.50	5.50	5.50	5.50	0.28	0.28	0.28	0.28
Chlorides	0.049	2440	2670	1750	1300	19.00	21.00	13.00	10.00	0.93	1.03	0.64	0.49
Total Iron	0.045	32	23	45.10	1.50	5.50	5.50	6.00	5.00	0.25	0.25	0.27	0.23
Summation	0.824									14.03	23.92	20.07	21.49
Derived LPI										17.03	29.02	24.36	26.08

Note: COD=chemical oxygen demand, BOD5= biological oxygen demand, TKN=total kjeldahl nitrogen, NH4-N=ammonia nitrogen, TCB= total colifom bacteria and TDS=total dissolve solid. All concentrations are in mg/L, except pH and total coliform bacteria unit (cfu/100ml) and NA= Leachate concentration not available.

Table 5 Calculation of LPI at 900 days after filling of MSW in landfill lysimeter

Pollutant variable	Variable weight, wi	Pollutant concentration, ci				Individual pollutant rating, pi				Overall pollutant rating, wipi			
		A1	A2	B	C	A1	A2	B	C	A1	A2	B	C
Chromium	0.064	0.055	0.04	0.07	0.065	5	5	5	5	0.32	0.32	0.32	0.32
Lead	0.063	0.126	0.265	0.55	0.476	5.5	6.5	7.5	7	0.3465	0.4095	0.4725	0.441
COD	0.062	852	4087	900	3300	33	63	36	55	2.046	3.906	2.232	3.41
Mercury	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
BOD5	0.061	345	437	490	510	12	15	16	19	0.732	0.915	0.976	1.159
Arsenic	-	NA	NA	NA	NA								
Cyanide	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
Phenol	-	NA	NA	NA	NA	-	-	-	-	-	-	-	-
Zinc	0.056	1.126	0.92	0.55	0.576	5	5	5	5	0.28	0.28	0.28	0.28
pH	0.055	6.51	7.27	6.45	7.21	7	5	7	5	0.385	0.275	0.385	0.275
TKN	0.053	233	686	402	300	8	22	12	8.5	0.424	1.166	0.636	0.4505
Nickel	0.052	0.055	0.04	0.07	0.065	5.5	5.5	5.5	5.5	0.286	0.286	0.286	0.286
TCB	0.052	122	178	181	104	49	56	56	48	2.548	2.912	2.912	2.496
NH4-N	0.051	196	165	208	198	19	17	20	20	0.969	0.867	1.02	1.02
TDS	0.05	5469	5437	6382	13879	12	12	14	26	0.6	0.6	0.7	1.3
Copper	0.05	0.43	0.07	0.07	0.4	6	5.5	5.5	6	0.3	0.275	0.275	0.3
Chlorides	0.049	2120.00	2376.00	1092.00	1187.00	15	17	9	10	0.735	0.833	0.441	0.49
Total Iron	0.045	5.9	3.2	4.5	2.6	5	5	5.5	5	0.225	0.225	0.2475	0.225
Summation	0.763									10.1965	13.2695	11.183	12.4525
Derived LPI										13.36	17.39	14.66	16.32

Note: COD=chemical oxygen demand, BOD5= biological oxygen demand, TKN=total kjeldahl nitrogen, NH4-N=ammonia nitrogen, TCB= total colifom bacteria and TDS=total dissolve solid. All concentrations are in mg/L, except pH and total coliform bacteria unit (cfu/100ml) and NA= Leachate concentration not available.

Table 6 Overall LPI in landfill lysimeter at varying operational condition

Days after filling	Operating condition of landfill lysimeter			
	A1	A2	B	C
7	27.36	36.91	33.52	32.16
14	26.48	36.13	34.04	32.37
21	26.31	35.72	33.96	30.71
30	28.78	32.13	32.12	32.85
40	25.29	30.30	29.20	29.19
50	17.26	31.23	32.99	31.22
60	19.70	30.75	27.87	30.06
70	25.05	30.61	28.75	29.25
90	21.69	28.55	24.63	30.07
120	17.03	29.02	24.36	26.08
140	24.23	30.22	28.18	28.54
160	22.40	27.34	24.00	24.47
190	24.67	29.72	28.17	28.30
220	24.23	29.27	28.05	28.36
245	18.33	25.70	22.31	24.83
280	24.04	28.71	26.78	27.60
300	17.89	24.07	22.61	25.41
320	23.29	28.40	26.60	27.48
380	16.80	23.20	20.39	24.52
390	14.74	20.63	17.75	18.79
400	14.69	20.53	17.43	18.54
415	14.15	20.53	17.31	18.17
500	15.41	22.25	22.64	22.61
520	14.81	22.02	18.72	21.00
545	17.19	24.10	21.30	18.54
580	17.25	20.59	18.51	19.99
610	14.33	20.73	19.22	20.36
630	18.47	19.48	19.61	17.87
660	13.86	20.06	17.48	19.48
715	17.18	18.24	18.18	14.54
750	16.72	17.64	17.15	12.94
800	13.36	17.39	14.66	16.32
850	16.37	15.27	16.03	12.61
900	14.52	13.82	13.79	12.07
Mean	19.53	25.33	23.48	23.74

valuation of Overall LPI

The predicted values of LPI in relation to the variation of lysimeter operating system such as detection (A1) and collection (A2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C, at the elapsed period 7- 900 days after filling of MSW in lysimeter is provided in in Table 6 and also in Figures 5, 6, 7 and 8. Figure depicts that derived LPI was deduced from 27.36-14.52 , 36.91-13.82, 33.52-13.79 and 32.16-12.07, for the detection (A1) and collection (A2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C, at the elapsed period 7- 900 day after filling of MSW in lysimeter, correspondingly. At the elapsed period 7, collection (A2) system of open dump lysimeter-A had the highest LPI (36.91) than the other counterparts that means detection (A1) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C (Figure 6). The highest LPI for the collection system of open dump lysimeter-A further indicates that MSW deposited in open dump lysimeter-A has not yet stabilized. This is also evident from the high BOD5 and COD values. The standard concentration for the disposal of treated leachate to inland surface water as per the Management and Handling Rules (The Gazette of Government of India 2000) should not exceed 2.0, 0.1, 250, 0.01, 30, 0.20, 0.20, 1.0, 5.0, 5.5-9.0, 100, 3.0, 50.0, 2100, 3.0 and 1000 mg/L for Cr, Pb, COD, Hg, BOD5, As, CN, Phenol, Zn, pH, TKN, Ni, NH3-N, TDS, Cu and Cl-, respectively and their corresponding overall LPI of 7.38. Moreover, the concentration of all the parameters of leachate in lysimeter-A, B and C, except Cr, Ni, Cu, pH and Zn exceed the permissible limits of treated leachate discharging into inland surface water (Table 3, Column 3). Here, it can be concluded that comparison of leachate characteristics with the standards set for the disposal of treated leachate verifies the fact that leachate generated fromlandfill lysimeter is highly contaminated and will have to be treated before discharge (so that the LPI comes below 7.38).

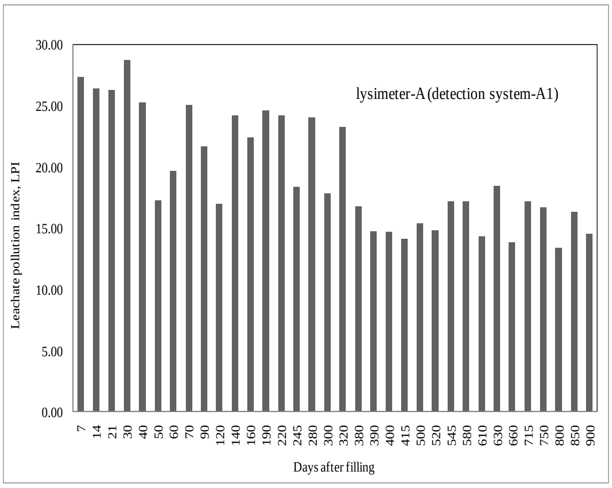

Figure 5 Overall LPI in detection system of open dump lysimeter-A

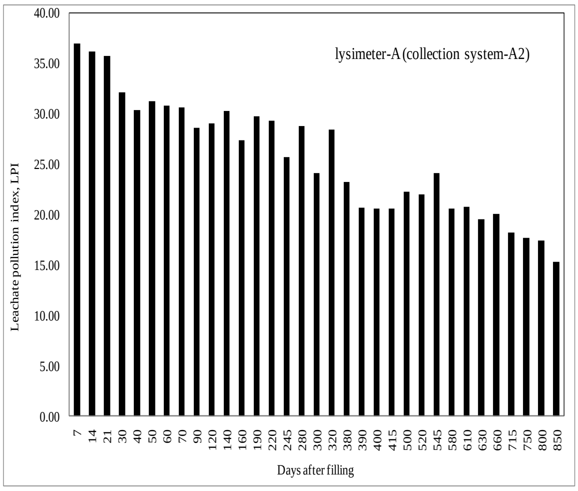

Figure 6 Overall LPI in collection system of open dump lysimeter-A

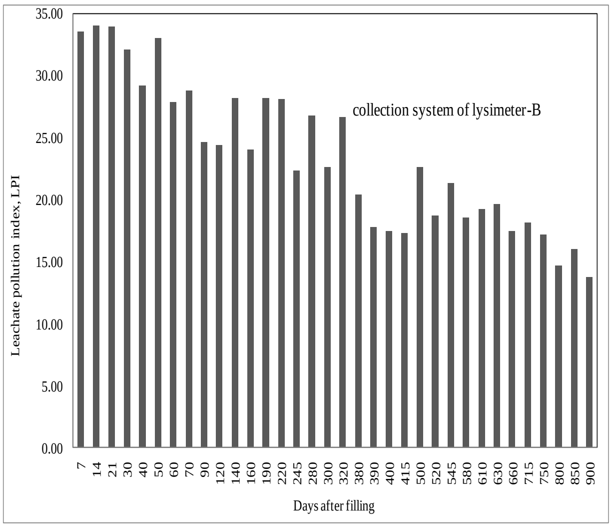

Figure 7 Overall LPI in collection system of sanitary landfill lysimeter-B

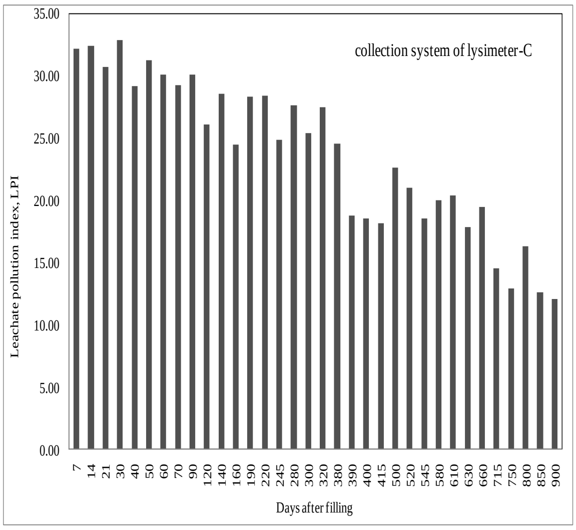

Figure 8 Overall LPI in collection system of sanitary landfill lysimeter-C

Result also reveals that leachate generated from entire landfill lysimeter is highly contaminated and LPI for all the operational condition of landfill lysimeter exceed the LPI (7.38) of treated leachate. The high LPI demands that leachate generated from the landfill lysimeter should be treated. Here, it is important to note that the decreasing trend of all the concentration of pollutant in leachate involved in LPI against the elapsed period was observed (Table 3, 4 and 5; Column 3) and it implies finally the decreasing pattern of LPI at varying elapsed period for entire lysimeter operating system. The lowest LPI (13.36) at elapsed period 800 day, for the detection (A1) system of open dump lysimeter-A (Figure 5) indicates the relatively lower contaminant potential. However, the individual contaminants shall meet the state discharge standards before discharge of leachate into any surface water body. Moreover, it can be depicted that comparatively the lowest LPI for the landfill lysimeter sites are attributable to low concentrations of heavy metals in the leachate (Table 3, 4 and 5; Column 3). Landfill age also plays an important role in the leachate characteristics and hence, influences the LPI value (Lo 1996).

The comparison of mean LPI values of lysimeter at four distinct operational condition of detection (A1) and collection (A2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C with the published results available in the literature for the same cases is provided in Figure 9. In the present study the evaluated mean LPI values were found 19.53, 25.33, 23.48 and 23.74 for the detection (A1) and collection (A2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C, respectively (Table 6). Moreover, a study conducted by Umar et al. (2010) for claculating LPI on four selected solid waste disposal sites in Malaysia named Pulau Burung landfills (PBLS) (sanitary landfill level III through leachate recirculation and controlled tipping), Ampang Jajar Landfill Site (AJLS) (semiaerobic closed landfill having no base liner), Kuala Sepetang Landfill Site (KSLS) (improved anaerobic landfill, natural marine clay and local soil are used as cover material for dumped waste with leachate collection pond) and Kulim Landfill Site (KLS). The derived LPI was found 23.45, 16.44, 21.77 and 19.50 for PBLS, AJLS, KSLS and KLS disposal sites, respectively. On the other hand, Kumar and Alappat (2003b) selected Okhla Sanitary landfill (OSL), New Delhi (no base liner or leachate collection and treatment systems) solid waste disposal site as a case study for calculating LPI and it was found 42.18. In addition, a study conducted by Kumar and Alappat (2005a) and selected four landfill named Ma Tso Lung (MTL) and Nagu Chi Wan (NCW) closed landfill sites as well as Pillar Point (PP) and Shuen Wan (SW) active landfill sites in Hong Kong for calculating LPI and it was found 45.01, 15.97, 36.48 and 39.04, respectively. Figure 9 depicts that the collection (A2) system of open dump lysimeter-A had the highest LPI value than that of other lysimeter operational condition. Here, it can be concluded that among the selected disposal sites, four landfill of OSL, MLT, PP and SW having more LPI due to their operational configuration than that of present lysimeter studies. Because the present study is a pilot scale laboratory experiment.

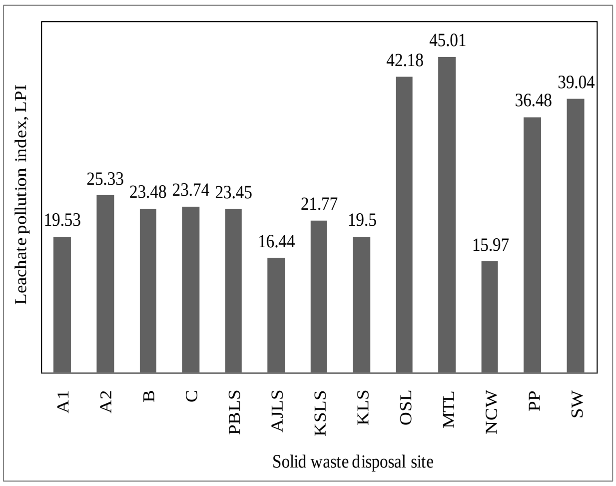

Figure 9 Comparison of the leachate contamination potential of Landfill lysimeter with the other researchers

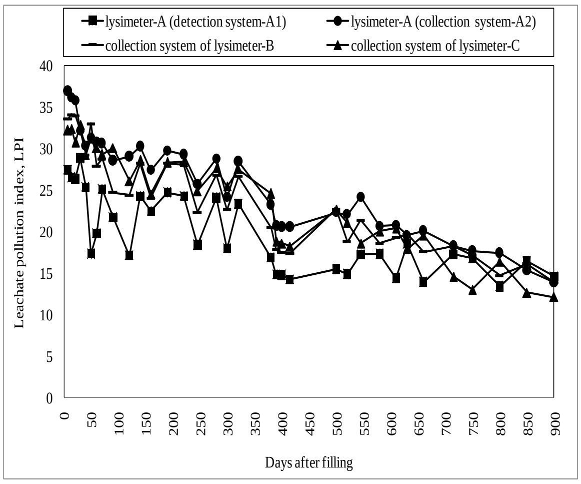

Figure 10 Variation of LPI with elapsed period of landfill lysimeter at varying operational condition

The decreasing trend of LPI at different operational condition of landfill lysimeter in relation to the increaing of days after filling of MSW is evident in Figure 10. It depicts that collection (A2) system of open dump lysimeter-A had the highest LPI, while, lowest for the detection (A1) system of lysimeter-A until the end of this study and provided in Figure 10. Moreover, collection system of open dump lysimeter-A shows the the highest individual and cumulative pollution rating and consequently the higher overall LPI because of the higher concentration of pollutant parameters in leachate (Figure 10). A study conducted by Truett (1975) and advocated that LPI is a planning index, specifically for decision-making, may be further generated, as one used by United States Environmental Protection Agency (USEPA) for planning MSW treatment project.

Moreover, Figure 10 also depicts that collection (A2) system of lysimeter-A had the highest LPI because of the highest concentration of pollutant parameters in leachate which was implied the highest individual and cumulative pollution rating and consequently the higher overall LPI. Here, it can be established that the variation of concentration of pollutant parameter in leachate in case of detection and collection system of open dump lysimeter-A may be occurred due to the providing of 400 mm thick CCL as a barrier between the detection and collection system of lysimeter-A. As the collection sysytem (A2) of open dump lysimeter-A was provided just below the MSW deposited in lysimeter-A and the followed detection sysytem (A1) was seperated with the 400mm thick CCL and this operational mode may be considered for the variation of pH between these two systems. So, it can be concluded that these dintinct variation of concentration of pollutant parameter in leachate finally implies the varied LPI between these two system.

In contrary, the collection (A2) system of open dump lysimeter-A had the highest LPI and it was continued untill the end of this study against the counterpart i,e. the collection system of sanitary landfill lysimeter-B and C also provided in Figure 10. It is important to note that the decreasing of LPI with the inceases of elapded period was occured due to the reduction of concentration of pollutant parameters involved in LPI. Moreover, Qasim and Chiang (1994) stated that variation of concentration of pollutant in MSW landfill leachate can be attributed to many interacting factors such as the composition and depth of MSW; decomposition and age of MSW; degree of compaction; landfill design and operation; liner (top and base) design and operation; MSW filling procedures; the availability of moisture content; available oxygen; rate of water movement (fluid addition) and temperature. However, it can be established that the variation of leachate concentration for the collection system of lysimeter-A and B may be occurred due to the simulation behavior of open dumping and providing the base liner in open dump lysimeter-A as well as sanitary landfill and providing the cap liner in lysimeter-B. In contrast, the variation of leachate concentration for the collection system of sanitary landfill lysimeter-B and C may be occurred due to the difference of landfill lysimeter-B and C in terms of thickness and compaction conditions of cap liner. So, it can be concluded that these dintinct variation of concentration of pollutant parameter in leachate finally implies the varied individual, cumulative pollutant rating and the overall LPI.

CONCLUSIONS

LPI is a good tool and it provides an evocative method of evaluating contamination potential of solid waste disposal sites. Moreover, LPI can be a useful tool to monitor the leachate trends over the lifetime of landfill site, and thus can help to take necessary decisions as deem fit. Result reveals that open dump lysimeter had the highest LPI than other counterparts and it also found that the clay liner system of landfill lysimeter protect the migration of contaminant thus producing the lowest value of LPI. Finally, it can be concluded that entire landfill lysimeter is highly contaminated, so proper treatment of leachate must be ensured before discharging into natural streams to protect the environment.

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Analysis of Errors Involved in the Estimation of Leachate Pollution Index Due to Nonavailability of Leachate Parameter of Landfill Lysimeter

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ABSTRACT

An environmental index known as leachate pollution index (LPI) to quantify and compare the leachate contamination potential of solid waste landfill in a given geographical area was developed and reported elsewhere. An important part of maintaining a solid waste landfill is managing the leachate through proper treatment methods designed to prevent pollution into surrounding ground and surface waters. Any assessment of the potential impact of a landfill on groundwater quality requires consideration of the components of the leachate most likely to cause an environmental impact as well as the source of concentration of those components. LPI is an environmental index and it is based on the concentration of 18 pollutants of the leachate and their corresponding significance. That means, for calculating the LPI of a landfill, concentration of these 18 parameters are to be known. However, sometimes the data for all the 18 pollutants included in the LPI may not be available to calculate the LPI. In this study, the possible errors involved in calculating the LPI due to the nonavailability of data are reported and analyzed by the author. The leachate characteristic data from pilot scale landfill lysimeter at KUET campus, Khulna, Bangladesh have been used to estimate and analysis these errors. Based on this study, it can be concluded that the errors may be high if the data for the pollutants having significantly high or low concentration are not available. However, LPI can be reported with a marginal error if the concentrations of the nonavailable pollutants are not completely biased.

Keywords: Landfill lysimeter, leachate pollutant, sub-index value, pollutant weight, lechate pollution index, error analysis

INTRODUCTION

The relative negative impacts of solid waste, dumping in a landfill of open condition are common practices and well known all over the world. So, it is necessary to save the environment and make it friendly against the mismanagement of MSW (Rafizul et at. 2011). Environmental degradation caused by the solid waste landfill in an uncontrolled conditions and dumps is a well-known fact. Air, water, and soil pollution from the uncontrolled dumping of waste in the dumping grounds and their ill effect on human health and environment have been reported in the past by many researchers. The most commonly reported danger to the human health from these landfills is from the use of groundwater that has been contaminated by leachate (Chian and DeWalle 1976; Kelley 1976; Lo 1996; Kumar et al. 2002). Landfill leachate is liquid that moves through or drains from a solid waste landfill due to the percolation of precipitation through open landfill or through cap of the completed site (Aziz et al. 2007). Moreover, landfill leachate is formed from the infiltration and passage of water through solid waste which results in a combination of physical, chemical and microbial processes that transfers pollutants from waste materials to the water (Jasper et al. 1985; Kjeldsen et al. 2002).In contrary, the characteristics of leachate are highly variable depending on the wastes deposited in the landfill, composition of wastes, moisture content, the particle size, the degree of compaction, sampling procedures, landfill design and operation, the hydrology of the site, the climate, and age of the fill and other site-specific conditions including landfill design and type of liners used, if any (Rafizul et al. 2011; Leckie et al. 1979). Leachate is the main medium for the contaminants transportation from the landfill to groundwater and surface water (Rowe 1995).

The term ‘landfill’ can be treated as synonymous to ‘sanitary landfill’ of solid waste, only if the latter is designed on the principle of waste containment and is characterized by the presence of a liner and leachate collection system to prevent ground water contamination. Sanitary landfill is one of the secure and safe facilities for the disposal of ssolid waste; however, it needs high standard of environment protection in the operation of landfill (Davis and Cornwell 1998). Modern landfills are often designed to prevent liquid from leaching out and entering the environment; however, if not properly managed, the leachate is at risk for mixing with groundwater near the site, which can have terrible effects (Chian and DeWalle 1976). Leachate can consist of many different organic and inorganic compounds that are typically either dissolved or suspended in the wastewater (Christensen and Kjeldsen 1995). The leachate may be virtually harmless or dangerously toxic, depending upon what is in the landfill (Leckie et al. 1979; Kouzeli-Katsiri et al. 1999).

A large number of environmental indices have been developed in last four daades. Various indices are developed to quantify the pollution or quality of water and air. Usually, the indices are formulated based on studies conducted by the indices developers or on the Delphi technique (Kumar and Alappat 2009). In an effort to develop a method for comparing the leachate pollution potential of various solid waste landfill sites in a given geographical area, an index known as Leachate Pollution Index (LPI) was formulated using Rand Corporation Delphi Technique (Kumar and Alappat 2003). The LPI is an increasing scale index, wherein a higher value indicates a poor environmental condition. The LPI has many possible applications including ranking of landfill sites, resource allocation, trend analysis, enforcement of standards, scientific research, and public information (Kumar and Alappat 2003). The LPI can be used to report leachate pollution changes in a particular landfill over time. The LPI can also be used to compare leachate contamination potential of different landfills in a given geographical area or around the world.The other potential applications of LPI include ranking of landfill sites based on leachate contamination potential, resource allocations for landfill remediation, enforcement of leachate standards, scientific research and public information (Kumar and Alappat 2003). The LPI is based on the concentration of 18 pollutants of the leachate and their corresponding significance. Hence, for calculating the LPI of a landfill, concentrations of these 18 parameters are to be known. However, it is possible that the data for all 18 pollutants included in the LPI are not available. The LPI calculated based on the available data is likely to involve some error and bias. In this study, the errors involved in calculating LPI due to the nonavailability of data are reported and analysis by the author. The leachate characteristic data from pilot scale landfill lysimeter at KUET campus, Khulna, Bangladesh have been used to estimate and analysis these errors.

METHODOLOGY ADOPTED

Leachate Pollution Index (LPI)

The formulation process and complete description on the development of the LPI has been discussed elsewhere (Kumar and Alappat 2003). The LPI represents the level of leachate contamination potential of a given landfill. It is a single number ranging from 5 to 100 (like a grade) that expresses the overall leachate contamination potential of a landfill based on several leachate pollution parameters at a given time.

LPI Variables and Their Weight

The 18 parameters chosen and their corresponding weights are as follows: chromium (Cr): 0.064; lead (Pb): 0.063; chemical oxygen demand (COD): 0.062; mercury (Hg): 0.062; biochemical oxygen demand (BOD_5): 0.061; arsenic (As): 0.061; cyanides (Cn): 0.058; phenolic compounds: 0.057; zinc (Zn): 0.056; pH: 0.055; total kjeldhal nitrogen (TKN):0.053; nickel (Ni): 0.052; total coliform bacteria (TCB): 0.052; ammonia nitrogen (NH_4-N): 0.051; total dissolved solids (TDS): 0.050; copper (Cu): 0.050; chlorides (Cl-): 0.048; and total iron (Fe): 0.044. The weight factor indicates the importance of each pollutant variable to the overall leachate pollution. The sum of the weights of all 18 parameters is one.

Variable Curves

The averaged sub-index curves for all the pollutant variables have been reported by Kumar and Alappat (2003).

Variable Aggregation

The weighted sum linear aggregation function was found to be the most suitable one for the calculation of LPI (Kumar and Alappat 2004) and is as follows:

$$LPI = \sum_{i=1}^{n} w_i p_i \qquad (1)$$

Where, where LPI is weighted additive leachate pollution index; w_i = the weight for the i^{th} pollutant variable;
p_i = the sub-index value of the i^{th} leachate pollutant variable, number of leachate pollutant parameters; n =18 and $\sum w_i = 1$.

However, when the data for all the leachate pollutant variables included in LPI are not available, the LPI can be calculated using the data set of the available leachate pollutants by the equation:

$$LPI = \frac{\sum_{i=1}^{m} w_i p_i}{\sum_{i=1}^{m} w_i} \tag{2}$$

variable. When, all the leachate parameters are unknown , m < 18 and $\sum_{i=1}^{m} w_i < 1$.

Errors Involved in Calculating LPI Due to Nonavailability of Data

To assess the errors involved in calculating LPI due to nonavailability of leachate data, a case study is taken up. Leachate samples from landfill lysimeter at KUET campus, Bangladesh were collected and analyzed in the laboratory and the concentration of pollutant parameters in leachate is provided in Table 1, to evaluate the error invlolved in calculation LPI due to nonavailability of data.

Table 1 Concentration of pollutant parameters in leachate of landfill lysimeter

SL	Leachate pollutant	Concentration*	Sub-index value (p_i)
1	Total chromium (Cr)	1.75	6.5
2	Lead (Pb)	0.68	9
3	Chemical oxygen demand (COD)	8425	70
4	Mercury (Hg)	0.4	59
5	Biochemical oxygen demand (BOD_5)	1398	35
6	Arsenic (As)	0.01	5.5
7	Cyanide (Cn)	1.3	10
8	Phenol compounds	4	8.5
9	Zinc (Zn)	1.3	5
10	pH	7.3	6
11	Total kjeldhal nitrogen (TKN)	3000	98
12	Nickel (Ni)	0.23	8
13	Total colifom bacteria (TCB)	8000	92
14	Ammonia nitrogen (NH_4-N)	1300	100
15	Total dissolved solid (TDS)	12540	28
16	Copper (Cu)	0.98	7
17	Chlorides (Cl-)	3597	30
18	Total Iron (Fe)	82	9.5

*All values in mg/L except pH and total coliform unit (cfu/100ml). * Average of 4 samples were taken between 1st September to 1st Novemver 2011.*

CASE STUDY

To estimate the possible errors involved in calculating LPI, due to the nonavailability of leachate data, two approaches have been made as

- Ignoring pollutant data based on weight factor and
- Ignoring pollutant data based on sub-index value.

The sub-index values of all the pollutant parameters in lechate from landfill lysimeter based on their concentrations are reported in Table 1. The subindex values have been derived from the subindex curves for all the parameters reported by Kumar and Alappat (2003). The LPI value based on these sub-index values has been calculated using Equation (1) and provided in the fifth column, Table 2. The LPI calculated based on these 18 parameters is considered to be the true LPI value of the landfill lysimeter.

Errors Introduced by Ignoring Pollutant Data Based on Weight Factor

In this approach, two options are discussed. In the first option, the data of the pollutants having low weight factors is ignored and in the second option, the data of the pollutants with high weight factors are assumed to be not available.

Removing Pollutants with Low Weight Factors

[1] In the first step, the concentration of the total iron, the parameter having lowest weight, is presumed to be unknown. Hence, by deleting the subindex value of total iron, the LPI value is derived by using Eq. (2). The derived LPI value is reported in the sixth column, Table 2.

[2] In the next step, the concentration of chlorides, the parameter having second lowest weight, is also presumed to be unknown in addition to the concentration of total iron. Again using Eq. (2), the LPI of the data set with 16 parameters is calculated and reported in the seventh column, Table 2.

[3] In a similar fashion, it is presumed that the concentrations of copper, total dissolved solids, ammonia nitrogen, total coliform bacteria, nickel, total kjeldhal nitrogen, pH, and zinc are also not known one by one in addition to the earlier unknown concentrations of the parameters. The derived LPI values considering concentration of 15, 14, 13, 12, 11, 10, 9, 8, 7, 6, 5 and 4 parameters are calculated and reported in columns 8, 9, 10, 11, 12, 13, 14, 15, 16, 17, 18 and 19 of Table 2, respectively.

[4] The percentage error introduced calculating LPI values, with respect to the LPI value when data are available for all 18 leachate pollutants, is also reported in the last row of respective columns of Table 2.

[5] The variation in LPI values with respect to the number of parameters considered in calculating LPI is provided in Figure 1. It also gives the percentage error introduced in calculating LPI values with respect to the number of parameters considered.

Removing Pollutants with High Weight Factors

A similar procedure was followed here starting with the parameterhaving the highest weight factor.

[1] In the first step, the concentration of the chromium, which has the highest weight factor, is presumed to be unknown. The LPI value, ignoring the subindex value of chromium, is calculated and reported in column 6, Table 3.

[2] Then, step by step it is presumed that the concentrations of the pollutants lead, COD, mercury, BOD_5, arsenic, cyanide, phenol, zinc, and pH are not known in addition to the earlier presumed unknown parameters. The LPI values so calculated are reported in Table 3.

[3] The percentage error in calculating LPI values, with respect to the LPI value when data for all 18 parameter are considered, is also reported in the last row of respective columns of Table 3.

[4] Figure 2 shows the variation in LPI values with respect to the number of pollutants considered in calculating LPI. It also gives the percentage error introduced in calculating LPI values with respect to the number of parameters considered.

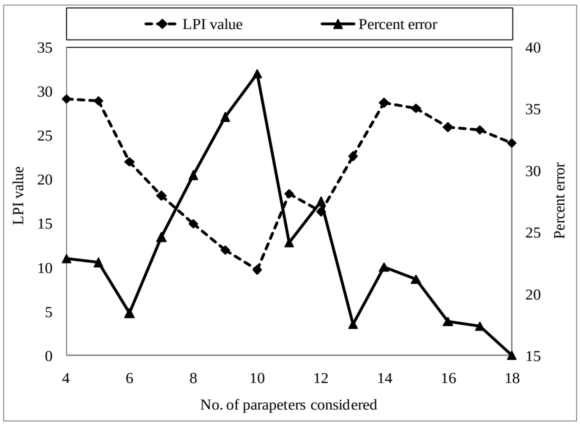

Figure 1 Variation of LPI and percent error when low weight parameters are ignored

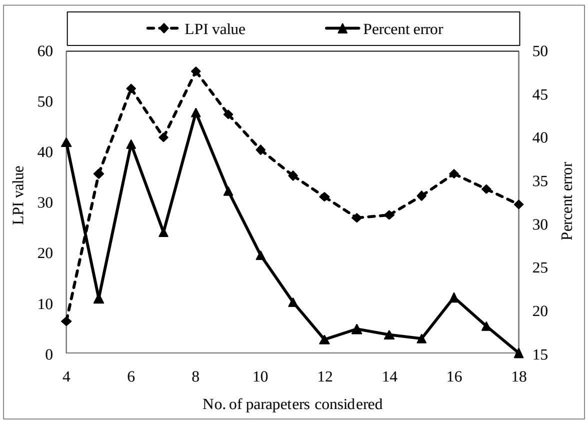

Figure 2 Variation of LPI and percent error when high weight parameters are ignored

Table 2. Estimating errors involved in calculating LPI values due to nonavailability of data (Parameters with low weight factors ignored)

Pollutant	Pollutant weight, w_i	Pollutant concentration, C_i	Subindex value, p_i	Derived LPI with considered leachate parameters (w_ip_i)														
				18	17	16	15	14	13	12	11	10	9	8	7	6	5	4
Cr	0.064	1.75	6.5	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416
Pb	0.063	0.68	9	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567
COD	0.062	8425	70	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34
Hg	0.062	0.4	59	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658
BOD_5	0.061	1398	35	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	-
As	0.061	0.01	5.5	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	-	-
Cn	0.058	1.3	10	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	-	-	-
Phenol	0.057	4	8.5	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	-	-	-	-
Zn	0.056	1.3	5	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28	-	-	-	-	-
pH	0.055	7.3	6	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	-	-	-	-	-	-
TKN	0.053	3000	98	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194	-	-	-	-	-	-	-
Ni	0.052	0.23	8	0.416	0.416	0.416	0.416	0.416	0.416	0.416	-	-	-	-	-	-	-	-
TCB	0.052	8000	92	4.784	4.784	4.784	4.784	4.784	4.784	-	-	-	-	-	-	-	-	-
NH_4-N	0.051	1300	100	5.1	5.1	5.1	5.1	5.1	-	-	-	-	-	-	-	-	-	-
TDS	0.05	12540	28	1.4	1.4	1.4	1.4	-	-	-	-	-	-	-	-	-	-	-
Cu	0.05	0.98	7	0.35	0.35	0.35	--	-	-	-	-	-	-	-	-	-	-	-
Chloride	0.048	3597	30	1.44	1.44	--	--	-	-	-	-	-	-	-	-	-	-	-
Iron	0.045	82	9.5	0.4275	--	--	--	-	-	-	-	-	-	-	-	-	-	-
Summation	1.000			32.238	31.810	30.370	30.020	28.620	23.520	18.736	18.320	13.126	12.796	12.516	12.032	11.452	11.116	8.981
Total weight				1.000	0.955	0.907	0.857	0.807	0.756	0.704	0.652	0.599	0.544	0.488	0.431	0.373	0.312	0.251
Derived LPI				32.238	33.309	33.484	35.029	35.465	31.111	26.614	28.098	21.913	23.522	25.648	27.915	30.701	35.628	35.781
Percent error				0.000	3.323	3.867	8.660	10.011	3.494	17.445	12.840	32.026	27.035	20.442	13.407	4.766	10.518	10.991

Note: All concentrations are in mg/L except for pH and total coliform bacteria (cfu/100mL)

Table 3. Estimating errors involved in calculating LPI values due to nonavailability of data (Parameters with high weight factors ignored)

Pollutant	Pollutant weight, w_i	Pollutant concentration, C_i	Subindex value, p_i	Derived LPI with considered leachate parameters ($w_i p_i$)														
				18	17	16	15	14	13	12	11	10	9	8	7	6	5	4
Cr	0.064	1.75	6.5	0.416	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Pb	0.063	0.68	9	0.567	0.567	-	-	-	-	-	-	-	-	-	-	-	-	-
COD	0.062	8425	70	4.34	4.34	4.34	-	-	-	-	-	-	-	-	-	-	-	-
Hg	0.062	0.4	59	3.658	3.658	3.658	3.658	-	-	-	-	-	-	-	-	-	-	-
BOD_5	0.061	1398	35	2.135	2.135	2.135	2.135	2.135	-	-	-	-	-	-	-	-	-	-
As	0.061	0.01	5.5	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	-	-	-	-	-	-	-	-	-
Cn	0.058	1.3	10	0.58	0.58	0.58	0.58	0.58	0.58	0.58	-	-	-	-	-	-	-	-
Phenol	0.057	4	8.5	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	-	-	-	-	-	-	-
Zn	0.056	1.3	5	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28	-	-	-	-	-	-
pH	0.055	7.3	6	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	-	-	-	-	-
TKN	0.053	3000	98	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194	-	-	-	-
Ni	0.052	0.23	8	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	-	-	-
TCB	0.052	8000	92	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	-	-
NH_4-N	0.051	1300	100	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	-
TDS	0.05	12540	28	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4
Cu	0.05	0.98	7	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35
Chloride	0.048	3597	30	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44
Iron	0.045	82	9.5	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275
Summation	1.000			32.238	31.822	31.255	26.915	23.257	21.122	20.786	20.206	19.722	19.442	19.112	13.918	13.502	8.718	3.618
Total weight				1.000	0.936	0.873	0.811	0.749	0.688	0.627	0.569	0.512	0.456	0.401	0.348	0.296	0.244	0.193
Derived LPI				32.238	33.997	35.801	33.187	31.050	30.700	33.152	35.511	38.519	42.635	47.660	39.993	45.613	35.727	18.744
Percent error				0.000	5.459	11.055	2.945	3.683	4.770	2.835	10.156	19.484	32.252	47.839	24.057	41.491	10.826	41.858

Note: All concentrations are in mg/L except for pH and total coliform bacteria (cfu/100mL)

Table 4. Estimating errors involved in calculating LPI values due to nonavailability of data (Parameters with highest sub-index values ignored)

Pollutant	Pollutant weight, w_i	Pollutant concentration, C_i	Subindex value, p_i	Derived LPI with considered leachate parameters ($w_i p_i$)								
				18	17	16	15	14	13	12	11	10
Cr	0.064	1.75	6.5	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416
Pb	0.063	0.68	9	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567
COD	0.062	8425	70	4.34	4.34	4.34	4.34	N.C	N.C	N.C	N.C	N.C
Hg	0.062	0.4	59	3.658	3.658	3.658	3.658	3.658	N.C	N.C	N.C	N.C
BOD_5	0.061	1398	35	2.135	2.135	2.135	2.135	2.135	2.135	N.C	N.C	N.C
As	0.061	0.01	5.5	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355	0.3355
Cn	0.058	1.3	10	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58
Phenol	0.057	4	8.5	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845
Zn	0.056	1.3	5	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28	0.28
pH	0.055	7.3	6	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33	0.33
TKN	0.053	3000	98	5.194	5.194	N.C	N.C	N.C	N.C	N.C	N.C	N.C
Ni	0.052	0.23	8	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416	0.416
TCB	0.052	8000	92	4.784	4.784	4.784	N.C	N.C	N.C	N.C	N.C	N.C
NH_4-N	0.051	1300	100	5.1	N.C	N.C	N.C	N.C	N.C	N.C	N.C	N.C
TDS	0.05	12540	28	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	N.C
Cu	0.05	0.98	7	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35
Chlorides	0.048	3597	30	1.44	1.44	1.44	1.44	1.44	1.44	1.44	N.C	N.C
Total Iron	0.045	82	9.5	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275
Summation	1.000			32.238	27.138	21.944	17.160	12.820	9.162	7.027	5.587	4.187
Total weight				1.000	0.949	0.896	0.844	0.782	0.720	0.659	0.611	0.561
Derived LPI				32.238	28.596	24.491	20.331	16.393	12.724	10.662	9.143	7.463
Percent error				0.000	11.296	24.031	36.933	49.149	60.529	66.926	71.638	76.851

Note: All concentrations are in mg/L except for pH and total coliform bacteria (cfu/100mL)
N.C= parameter not considered

Table 5. Estimating errors involved in calculating LPI values due to nonavailability of data (Parameters with lowest sub-index values ignored)

Pollutant	Pollutant weight, w_i	Pollutant concentration, C_i	Subindex value, p_i	Derived LPI with considered leachate parameters ($w_i p_i$)								
				18	17	16	15	14	13	12	11	10
Cr	0.064	1.75	6.5	0.416	0.416	0.416	0.416	N.C	N.C	N.C	N.C	N.C
Pb	0.063	0.68	9	0.567	0.567	0.567	0.567	0.567	0.567	0.567	0.567	N.C
COD	0.062	8425	70	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34	4.34
Hg	0.062	0.4	59	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658	3.658
BOD_5	0.061	1398	35	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135	2.135
As	0.061	0.01	5.5	0.3355	0.3355	N.C	N.C	N.C	N.C	N.C	N.C	N.C
Cn	0.058	1.3	10	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58	0.58
Phenol	0.057	4	8.5	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	0.4845	N.C	N.C
Zn	0.056	1.3	5	0.28	N.C	N.C	N.C	N.C	N.C	N.C	N.C	N.C
pH	0.055	7.3	6	0.33	0.33	0.33	N.C	N.C	N.C	N.C	N.C	N.C
TKN	0.053	3000	98	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194	5.194
Ni	0.052	0.23	8	0.416	0.416	0.416	0.416	0.416	0.416	N.C	N.C	N.C
TCB	0.052	8000	92	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784	4.784
NH_4-N	0.051	1300	100	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1	5.1
TDS	0.05	12540	28	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4
Cu	0.05	0.98	7	0.35	0.35	0.35	0.35	0.35	N.C	N.C	N.C	N.C
Chlorides	0.048	3597	30	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44	1.44
Total Iron	0.045	82	9.5	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275	0.4275
Summation	1.000			32.238	31.958	31.622	31.292	30.876	30.526	30.110	29.626	29.059
Total weight				1.000	0.944	0.883	0.828	0.764	0.714	0.662	0.605	0.542
Derived LPI				32.24	33.85	35.81	37.79	40.41	42.75	45.48	48.97	53.61
Percent error				0.00	5.01	11.09	17.23	25.36	32.62	41.09	51.90	66.31

Note: All concentrations are in mg/L except for pH and total coliform bacteria (cfu/100mL)
N.C= parameter not considered

Table 6: Estimating errors involved in calculating LPI values due to nonavailability of data (Parameters with highest and lowest sub-index values ignored simultaneously)

Pollutant	Pollutant weight, w_i	Pollutant concentration, C_i	Subindex value, p_i	Derived LPI with considered leachate parameters (w_ip_i)				
				18	16	14	12	10
Cr	0.064	1.75	6.5	0.416	0.416	0.416	0.416	N.C
Pb	0.063	0.68	9	0.567	0.567	0.567	0.567	0.567
COD	0.062	8425	70	4.34	4.34	4.34	N.C	N.C
Hg	0.062	0.4	59	3.658	3.658	3.658	3.658	N.C^a
BOD$_5$	0.061	1398	35	2.135	2.135	2.135	2.135	2.135
As	0.061	0.01	5.5	0.3355	0.3355	N.C	N.C	N.C
Cn	0.058	1.3	10	0.58	0.58	0.58	0.58	0.58
Phenol	0.057	4	8.5	0.4845	0.4845	0.4845	0.4845	0.4845
Zn	0.056	1.3	5	0.28	N.C	N.C	N.C	N.C
pH	0.055	7.3	6	0.33	0.33	0.33	N.C	N.C
TKN	0.053	3000	98	5.194	5.194	N.C	N.C	N.C
Ni	0.052	0.23	8	0.416	0.416	0.416	0.416	0.416
TCB	0.052	8000	92	4.784	4.784	4.784	4.784	4.784
NH$_4$-N	0.051	1300	100	5.1	N.C	N.C	N.C	N.C
TDS	0.05	12540	28	1.4	1.4	1.4	1.4	1.4
Cu	0.05	0.98	7	0.35	0.35	0.35	0.35	0.35
Chlorides	0.048	3597	30	1.44	1.44	1.44	1.44	1.44
Total Iron	0.045	82	9.5	0.4275	0.4275	0.4275	0.4275	0.4275
Summation	1.000			32.238	26.858	21.328	16.658	12.584
Total weight				1.000	0.893	0.779	0.662	0.536
Derived LPI				32.24	30.08	27.38	25.16	23.48
Percent error				0.00	6.71	15.07	21.94	27.17

Note: All concentrations are in mg/L except for pH and total coliform bacteria (cfu/100mL)
N.C= parameter not considered

Introduced by Ignoring Pollutant Data Based on Sub-indexValue

In this approach, three scenarios were considered as reveals:

[1] Firstly, it was presumed that data for one parameter having the highest sub-index value are not available and then calculate LPI. Next it is presumed that the data for two sub-indices having the highest sub-index values are not available and so on for three, four, fifth to eight parameters. From Table 4, column 4, it is observed that the pollutants having the highest sub-index values are NH$_4$-N, TKN, TCB, COD, Hg, BOD$_5$ chloride and TDS with sub-index values of 100, 98, 92, 70, 59, 35, 30 and 28 respectively. To start with, it is presumed that data for NH$_4$-N are not available. Based on this assumption, the LPI value is calculated using Eq. (2), and reported in column 6, Table 4. In the next step it is presumed that data for TKN are also not available in addition to NH$_4$-N. The LPI value based on this assumption is calculated using Eq. (2) and reported in column 7, Table 4. Similarly, it is presumed that data for three and then four pollutants are not available, and the corresponding LPI values are calculated and reported in columns 8 and 9 of Table 4. Moreover, in the similar fashion, it is presumed that data for fifth to eight pollutants are not available, and the corresponding LPI values are calculated and reported in columns 10 to 13 in Table 4. The percentage error introduced in calculating these fourteen LPI values is also calculated and reported in the last row of respective columns in Table 4 and the results are shown in Figure 3.

[2] Then it is presumed that the data for one parameter having the least sub-index value are unknown and then LPI calculated. Subsequently it is presumed that, data for two, three, four and eight parameters having the lowest sub-index values are not available. From column 4 of Table 5, it is observed that the parameters having the lowest sub-index values are Zn, As, pH, Cr, Cu, Ni, phenolic compound and Pb with sub-index values of 5, 5.5, 6, 6.5, 7, 8, 8.5 and 9 respectively. The above procedure is repeated for calculating LPI values (reported in Table 5). The percentage error introduced in calculating LPI for each presumption is also calculated and reported in respective columns of Table 5 and the results are shown in Figure 4.

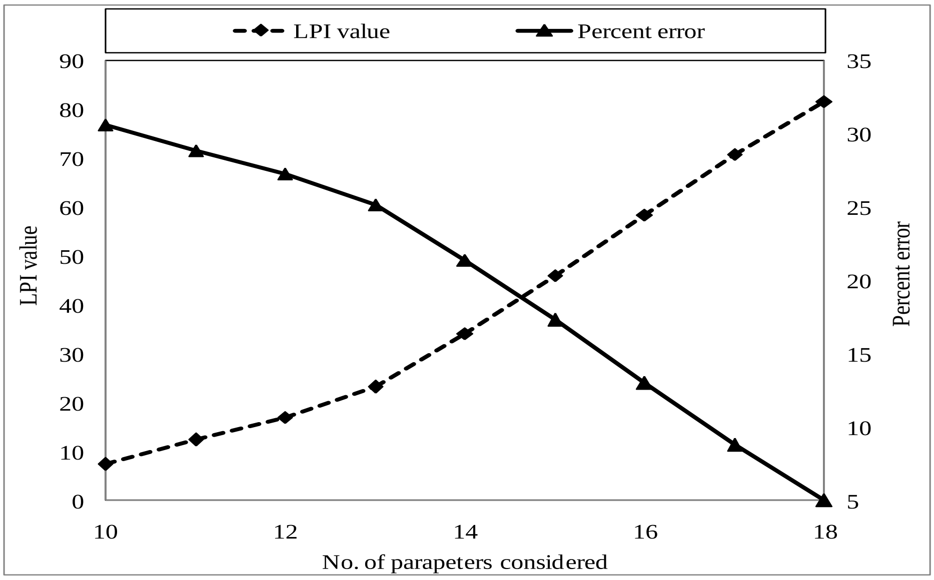

Figure 3 Variation of LPI and percent error when parameters with highest sub-index values ignored.

[3] After that it is presume that the data for two parameters, having the highest and lowest sub-index value are unknown simultaneously. The parameter having the highest sub-index value is NH_4-N and the one having the lowest sub-index value is Zn. The LPI for this is calculated and reported in column 6, Table 6. Subsequently it is presume that data for four parameters: two parameters with highest sub-index values (NH_4-N and TKN) and two parameters with lowest sub-index values (Zn and As) are not available. The LPI for this presumption is calculated and reported in column 7, Table 6. Moreover, it is presume that data for six parameters: three parameters with highest sub index values (NH_4-N, TKN and COD) and three parameters with lowest sub-index values (Zn, As and pH) are not available. The LPI for this presumption is calculated and reported in column 8, Table 6. Subsequently, it is presume that data for eight parameters: four parameters with highest and four with lowest sub index values are not available. The LPI for this presumption is calculated and reported in column 9, Table 6. The percent error introduced in calculating LPI values is reported in the last row of respective columns in Table 6 and the results are shown in Figure 5.

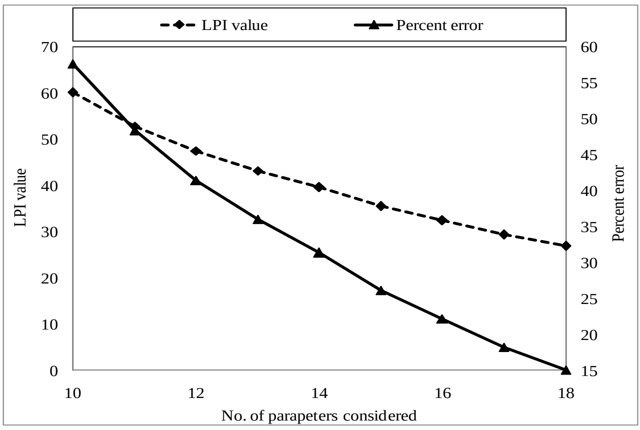

Figure 4 Variation of LPI and percent error when parameters with lowest sub-index values ignored

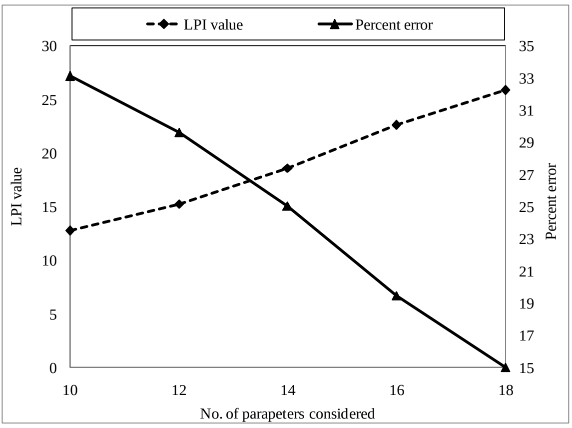

Figure 5 Variation in LPI due to nonavailability of data parameters with highest and lowest sub-index values ignored simultaneously

RESULTS AND DISCUSSIONS

Errors Introduced by Ignoring Pollutant Data Based on Weight Factor

Based on Table 2 and Figure 1, it can be depicted that the error introduced in calculating LPI is 3.32%, when concentration of one parameter, i.e., total iron (pollutant with lowest weight factor) is not considered. It depicts that error increases to 3.87% when concentration of two parameters, total iron and chlorides, is unknown. Then the error also increases to 8.66% when data for three parameters, total iron, chlorides, and copper, is not known. The error introduced in LPI is the highest, i.e., 32.03%, when data of eight parameters is not considered. After this, the error decreases with the increase in the number of missing parameters. The error is just 4.77% when only six parameters are considered (that is when data for twelve parameters are not considered).

Similarly from Table 3 and Figure 2, it is observed that the error is 5.46%, when data for one parameter that is chromium (pollutant with highest weight factor) is not considered in calculating LPI. The error increases to 11.06% when data for two pollutants, chromium and lead, are ignored in calculating LPI. The error is highest, i.e., 47.84%, when data for the ten pollutants are ignored. But the percent error dips to 24.06% when eleven parameters are not considered in calculating the LPI value. This leads to the conclusion that the error involved in calculating LPI does not vary with the number of parameters considered and the variation is erratic. The erratic behavior in the error introduced in the LPI is due to the fact that the parameters ignored while calculating LPI had significantly different subindex values with respect to the overall LPI of landfill lysimeter.

Errors Introduced by Ignoring Pollutant Data Based on Subindex Value

The error introduced is highest, i.e., 76.85%, when data for the eight parameters having the highest subindex values are not considered (Table 4, column 13) and it was found 71.64% when data for the seven parameters having the highest subindex values are not considered. However, it may reduce to 24.03% and 11.30 when data for the two and one parameters having the highest subindex values, respectively are not considered. Moreover, the errors introduced due to nonconsideration of data of one parameters having the lowest subindex values are 5.01% (Table 5, column 6) and it rises gradually to 66.31% (Table 5, column 13) when data for the eight parameters are not considered.

Here, it is important to note that derived LPI are lower than the true LPI value in the case when pollutants with high sub-index values are ignored. On the contrary, the derived LPI are higher than the true LPI when data for the pollutants with low sub-index values are ignored. Hence, the results obtained by ignoring data for the pollutants with high subindex values produce falsified results, leading to a false sense of security, indicating a relatively more polluted environment as less polluted.

But in the case when data for the pollutants with low subindex values are ignored, distended results are obtained and the results will raise an unnecessary alarm by indicating a comparatively less polluted environmental situation to be more contaminated. Based on this discussion, it is possible to conclude that the errors involved in LPI values are high and dangerous when the data for the pollutants having high subindex values are not available as compared to the scenario when data for the parameters having low subindex values are not available. The error involved in LPI values is low when data for the pollutants having highest and lowest subindex values are not considered simultaneously.

CONCLUSIONS

Result reveals that maximum error (32.03%) is introduced in calculating LPI when the data for the eight low weight parameters are not considered, but the error is as low as 3.32% when data for one parameter are not considered. Similarly the error involved in calculating LPI is maximum (47.84%) when data for ten high weight parameters are ignored, but the error involved is low (2.95%) when data for three high weight factors are ignored. Here it can be concluded that the errors introduced in calculating LPI values are not at all related to the number of parameters whose concentrations are not known. From this it can be concluded that LPI is more reliable and accurate as a larger number of parameters are available in its formulation. In contrary, the error introduced in calculating LPI is more sensitive when data for the parameters having high sub-index values are not considered as the derived LPI values are lower than the true LPI value and produce vague results. Finally, it can be concluded that errors introduced in calculating LPI are marginal when the data of the parameters having highest and lowest sub-index values are not considered simultaneously.

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Heavy Metal Leaching in Leachate Generated from Municipal Solid Waste of Landfill Lysimeter

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ABSTRACT

In this study, heavy metal leaching from open and sanitary landfill lysimeter test cells at KUET campus, Khulna, Bangladesh for municipal solid waste (MSW) has been investigated. To these endeavor, leachate samples from MSW deposited in landfill lysimeter were collected periodically and hence characterized in terms of metal concentrations of Calcium (Ca), Potassium (K), Sodium (Na), Magnesium (Mg) and Iron (Fe) as well as heavy metal concentrations of Cadmium (Cd), Chromium (Cr), Copper (Cu), Lead (Pb), Nickel (Ni), Zinc (Zn) and Manganese (Mn). Three different situations of sanitary landfill were considered here. Both the open dump and sanitary landfill conditions having a base liner and two different types of cap liner were simulated. Based on evaluated results, it can be depicted that the metal concentrations which were comparatively higher in leachate of open dump lysimeter were Ca and K, however, the heavy metal concentrations of Cd, Cu, Zn and Mn, and those apparently lower were metals of Na, Mg and Fe as well as heavy metals of Cr, Pb and Ni. However, significant release of heavy metals under open dump lysimeter was observed compared to sanitary lysimeter. Moreover, meaningful correlation between dissolved organic carbon (DOC) and leaching of Cu and Pb was another interesting observation. Result reveals that lysimeter operational mode had direct effect on leachate quality. Finally, it can be concluded that the knowledge of leachate quality will be useful in planning and providing remedial measures of proper liner system in landfill design and leachate treatment.

INTRODUCTION

Municipal solid waste (MSW) landfill leachate is considered one of the types of wastewater with the greatest environmental impact if the leachate is not controlled properly (Rafizul et al. 2009b). The leachate from MSW landfills are highly contaminated and have higher concentration of organic as well as inorganic pollutants and toxic substances such as metals (Kettunen et al. 1998). El-Fadel et al. (2002) reported that, the composition of landfill leachate can exhibit considerable spatial and temporal variations depending upon the site, management practices, refuse characteristics (i.e., composition and age), and internal landfill processes (hydrolysis, adsorption, speciation, dissolution, ion exchange, re-dox reactions, precipitation etc.). Therefore, it is difficult to generalize the chemical composition that a leachate posses at a particular time of sampling.

Landfill leachate is linked to the high concentrations of several pollutants. Moreover, it contains pollutant that can be divided into four main groups: dissolved organic matter; inorganic compounds, such as ammonium (NH_3), Ca, Mg, Na, K, iron (Fe), sulphates (SO_4), chlorides (Cl^-); heavy metals, such as Cd, Cr, Cu, Pb, Ni, Zn, Mn and xenobiotic organic substances (Andreottola et al. 1990). Landfill leachate might contain heavy metals in considerable concentrations among many other constituents. Heavy metals may constitute an environmental problem, if the leachate migrates into surface water or groundwater, or a treatment issue where the leachate is collected and treated prior to discharge. Thus, during recent decades, monitoring of heavy metals in landfill leachate has commonly been prescribed by the authorities and routinely performed by landfill operators (Baun et al. 2004). In the acidic conditions of a landfill, however, metals such as Cd, Cu, Fe, Mn, and Pb can dissolve and migrate with leachate. Other chemical reactions in a landfill can also change the state of metals, allowing them to attach to other particles and travel with leachate. Four processes have been reported to control heavy metal concentrations in landfill leachate such as complexation, oxidation–reduction reactions, sorption and precipitation (Christensen et al. 2001).

In South and South East Asia more than 90% of MSW is disposed of in open dumps. The numerous measures of MSW are available such as landfills, composting, sanitary landfill digestion, combustion or incineration, and gasification. The MSW dumps, which are common practice, are normally left uncovered, not compacted and daily cover system and there are two options for MSW dumping all over the world, one is cruded landfill (open dumping) another is sanitary landfill (Alamgir et al. 2005). Open dumping practice takes up lots of land, leads to serious pollution of its surrounding specially in ground water and exposed to scavengers and disease vectors (Demir et al. 2004). Shortage of covers, lack of leachate collection and treatment, inadequate compaction, poor site design, and rag pickers invasion are common at these open dumps. Growing concern about public health, environmental quality and risks associated with the existing dumps make it almost impossible to site new landfills in many parts of the world.

Controlled dumps are considered to be the first level of improvement to the open dumps for moving to engineered landfills. In fact, the controlled dumps are designed and constructed to eliminate problems associated with open dumps. For example; controlled dumps have the following features to minimize contamination of ground water from landfill leachate, a major concern with open dumps (Fang et al. 1999): (i) bottom liner system with leachate collection and removal systems; (ii) final cover systems to minimize infiltration of precipitation that contributes to the production of leachate. Considering this issue, it is necessary to develop strategies to design and operate simple landfills, which are in-transition between open dumps and sanitary landfill (Visvanathan et al. 2002). To this endeavor, three sanitary landfill lysimeter-s test set-up were designed and hence constructed at KUET campus, Khulna. However, the metal concentrations in terms of Ca, K, Na, Fe, Mg as well as heavy metal concentrations of Cd, Cr, Ni, Pb, Zn, and Cu in leachate originating from these lysimeter operation were monitored and hence compared with the results of other researchers to depict the validity of this study.

MATERIALS AND METHODS

Experimental Set-up of Landfill Lysimeter at KUET Campus

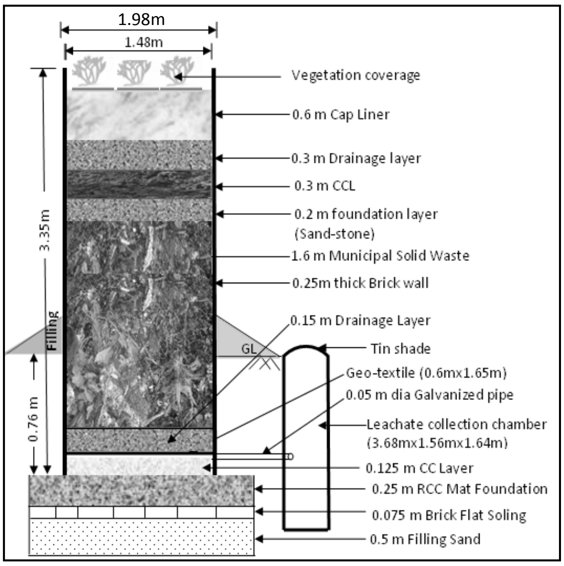

Figure 1. Schematic diagram of reference cell for landfill lysimeter design

The landfill lysimeter test facilities were set-up in the geo-environmental research station at the backyard of Civil Engineering Building at KUET campus, Bangladesh. In this study three different situations of landfill were considered in the lysimeter test set-up. An open dump lysimeter-A having a base liner and sanitary landfill lysimeter-B and C at two different types of cap liner were simulated. The lysimeter-A represents the present practice of open dumping of MSW as well as lysimeter-B and C similar to a pilot scale sanitary landfill (PSSL) at Rajbandh, Khulna, Bangladesh. Refer to Figure 1, as the technical details of reference cell for the construction of desired three lysimeters. In reference cell, the deposited MSW mainly consists of 93(w/w) organic (food and vegetables), 3(w/w) of plastic/polythene and 2(w/w) of leather/rubber, 1(w/w) of animal bone and rubber/leather as well as 1(w/w) of rope/straw and egg pill. The organic and moisture content of deposited MSW were found 52% and 65%, respectively, and the total volume was 2.80m^3 (height 1.6m) with a manual compaction to achieve the unit weight of 1,064kg/m^3. At the bottom of reference cell, a concrete layer of 125 mm thick was provided then the lysimeters were filled with stone chips (diameter 5-20 mm) and coarse sand (diameter 0.05-0.40mm) to the height of 15 cm of each to ensure uniform and uninterrupted drainage. The clay used as compacted clay liner (CCL) is characterized as the percentages of sand, silt and clay of 10, 56.6 and 33.4, respectively, while, optimum moisture content and maximum dry density of 18% and 16kN/m^3, respectively, and coefficient of permeability of 1.90x10^{-7}cm/sec.

The catagory of lysimeter, operational condition and the total weight of MSW deposited in each lysimeter are given in Table 1. In open dump lysimeter-A, a compacted clay liner (CCL) of 400mm thick was provided as a barrier between the leachate detection (A_1) and collection (A_2) system. The three landfill lysimeters were constructed using the brick wall of 250mm thick having outer and inner diameter of approximately 1.98m and 1.48m, respectively, with a total height of 3.35m, resting on a 250mm thick of RCC mat foundation resting at a depth of 760mm below the existing ground surface. The landfill lysimeter was plastered both the inner and outer sides with two coatings of waterproofing agent to avoid leakages and corrosion due to acidic environment. Further, the sanitary landfill lysimeters (B and C) consist of landfill gas (LFG) collection system above the MSW and leachate recirculation system below the MSW. At the base of each landfill lysimeter after placing the perforated leachate collection pipe, a geo-textile sheet having 0.60m wide and 1.65m length was placed to avoid rapid clogging by the sediments from landfill lysimeter. A leachate collection tank (3.68x1.56x1.64m) accommodating four separate leachate discharge pipes in the temporary collection and storage containers, were constructed using 250mm thick brick wall.

Table 1 Specifications, operational conditions of lysimeter used to simulate different landfill conditions

lysimeter	*Operating condition*	*Refuse (kg)*	*Liner specification*	*Simulation*
A	Open dump lysimeter with leachate detection (A_1) system	2860	400mm thick CCL as a barrier between detection and collection system of lysimeter-A	present practice of open dumping
	Open dump lysimeter with leachate collection (A_2) system			
B	Sanitary landfill lysimeter with gas measurement and leachate recirculation system	2985	Cap liner-I (300mm thick CCL)	applicability of designed top cover
C		2800	Cap liner-II (900mm thick natural top soil)	

Analytical Method of Leachate Characterization

The dissolved organic carbon (DOC) was measured in the laboratory by combustion (NPOC) with a Shimadzu TOC-5000 instrument as per the standard method APHA (1998) provided in Table 2. However, the Ca, Na, K and Mg ions were determined using flame atomic absorption spectrophotometer (VARIAN; AA/2400) with proper standard calibrations. In addition, Heavy metals viz., Cu, Cr, Cd, Ni, Pb, Mn, Fe and Zn were analyzed using spectrophotometer (HACH; DR/2400), according to standard methods APHA (1998).

RESULTS AND DISCUSSIONS

Metals Concentration

According to a statistical evaluation by Kylefords and Lagerkvist (1997), the concentration of metals are expected to reduce as the leachate changes from acidogenic to methanogenic, concurrent with a decrease in redox potential and an increase in pH. This expected decreasing trend is obvious in the Fe, Ca and Mg and but less defined in the Na and K, because of the sorption and precipitation that occur at higher pH are effective for Fe, Ca and Mg, and have a minor effect for Na and K (Christensen et al. 2001). The rate of Fe oxidation at pH level below 6.0 is increased by the presence of certain inorganic catalysts of the action of micro-organisms. The Fe concentration also imparts a test to water, which is detectable at very low concentration (Flayhammar et al. 1999). However, the changes of metal in terms of Ca, K, Na, Mg and Fe with in relation to the increasing of elapsed period provided depicted in Table 1 and hence discussed in followings.

Calcium Concentration

Calcium and Magnesium concentrations are the two important ions in the evaluation of buffering capacity of leachate during the waste degradation in landfill (Karthikeyan et al. 2004). The Ca concentration has a tendency of complex formation with $HCO3^-$ and dissolved organic compounds and dissolution reactions involving calcite ($CaCO_3$) and may be siderite ($FeCO_3$) (Christensen et al. 2001). Thus, the dissolution process is closely linked to the dissolved carbonate compounds.

The variation of Ca concentration in leachate with in relation to the increasing of elapsed period from MSW deposition in open dump lysimeter-A and sanitary landfill lysimeter--B and C is evident in Figure 2. The Figure 2 reveals that Ca ranging from 253.5 to 789.9 mg/L, 217.9 to 664.2 mg/L, 153.0 to 578.0 mg/L and 140.0 to 664.4 mg/L, for leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter--B and C, respectively. Here, it is interesting to note that the highest Ca was measured in leachate detection (A_1) system of open dump lysimeter-A with 789.9 mg/L, whereas, the lowest of 140 mg/L for sanitary landfill lysimeter-B. It may be established that the variation of Ca regarding to entire lysimeter operation systems may be occurred due to provide of 400 mm thick CCL as a barrier between leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, simulation behaviour of open dumping and base liner as well as sanitary landfill and cap liner of open dump lysimeter-A and sanitary landfill lysimeter-B as well as the difference of sanitary landfill lysimeter-B and C in terms of thickness and compaction conditions of cap liner.

Table 2 Concentration of heavy metal and metal in different leachate

Parameters (mg/L)	Jensen et al. (1999)	Kjeldsen et al. (2001)	Kalyuzhnyi et al. (2004)	Sinan et al. (2006)	WasteSafe (2010)	Diaz et al. (1996)	Results in the present study			
							A_1	A_2	B	C
Metal concentrations										
Calcium (Ca)	-	-	-	120-980	-	5-4080	253.5-789.9	217.9-664.2	153.0-578.0	140.0-664.4
Potassium (K)	-	-	-	1600-5000	-	2.8-3770	855.9-1967	1102.1-2589	1008.6-1764.3	567.2-1956.7
Sodium (Na)	-	-	-	800-5100	-	0-7700	1267.9-2424.9	1251.6-2369.6	1256.3-2764.3	1011.9-2409.1
Magnesium (Mg)	--		-	-	-	0.06-1400	147.25-589.9	127.15-564.2	228.4-605.9	156-594.2
Iron (Fe)	-	-	4.08–170.4	40-100	-	0.2- 5500	1.8-70.0	3.4-82.6	3.6-91.2	1.4-72.0
Heavy metal concentrations										
Cadmium (Cd)	0.0002-0.0036	0.0068	0.0008–0.024	0.04-3.0	0.006-0.009	-	0.04-0.265	0.04-0.168	0.04-0.178	0.04-0.105
Nickel (Ni)	0.028-0.084	0.13	-	0.1-4.8	0.03-0.11	-	0.04-0.075	0.04-0.055	0.045-0.09	0.04-0.07
Zinc (Zn)	0.085-5.31	0.67	0.86–28.8	0.1-21	0.04-0.75	0-1000	0.25-1.267	0.15-0.97	0.10-0.55	0.10-0.576
Copper (Cu)	0.002-0.0034	0.07	0.04–0.24	0.1-3.3	0.04-1.1	0-9.9	0.04-0.98	0.04-0.97	0.04-0.76	0.04-0.60
Chromium (Cr)	0-0.019	0.076	-	0.01-7.8	0.027-0.038	-	0.01-0.07	0.01-0.056	0.023-0.03	0.01-0.025
Manganese (Mn)	-	-	-	-	0.4-3.9	16.5-15000	0.80-6.0	2.5-21.2	0.50-19.3	0.80-8.0
Lead (Pb)	0-0.016	0.07	0.038–0.1	0.01-0.11	0.03-0.086	-	0.12-0.32	0.179-0.397	0.25-0.78	0.10-0.476

On the other side, Ca concentration was decreased rapidly below 400 mg/L after day 300 in both the cases of leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A. However, causes a slow decrease in Ca concentration for both the cases of sanitary landfill lysimeter-B and C, and markedly decreased around 200 mg/L after day 650. Christensen et al. (2001) reported that as a result of the rapidly dissolution process, leachate generated from MSW deposited in landfill that has high alkalinity values (~10,000 mg/L $CaCO_3$) and the Ca concentration was decreased. However, the slow increase of pH and the slowly generation of the dissolution process due to sanitary landfill condition, causes a slow decrease in Ca concentration. Moreover, a study conducted by Sinan et al. (2006) reported that in case of both the open dump landfill rectors (A_1 and A_2), the Ca concentration was decreased below 100 mg/L after day 100 of operation and markedly slow decrease was observed for both the sanitary landfill reactors AN_1 and AN_2, with 100 mg/L after 600 days of landfilling in reactors.

So, here, it was to a great extent to state that the findings in this study are in agreement with the postulation given by Christensen et al. (2001) and Sinan et al. (2006) because the alkalinity was found as round (~10,000 mg/L $CaCO_3$), markedly increasing trend of pH was occurred at the operating day 100 as well as resemblance of present lysimeter practices with the simulated by Sinan et al. (2004), for both the leachate detection (A_1) collection system (A_2) of open dump lysimeter-A and providentially, in this case, the Ca concentration was decreased markedly. Finally, in can be concluded that the Ca concentration in terms of minimum and maximum with respect to entire lysimeter operation systems shown in Table 1, the maximum value exists within the range given by Sinan et al. (2006) and also Diaz et al. (1996), however, the minimum concentration exceed the minimum limit they were stated.

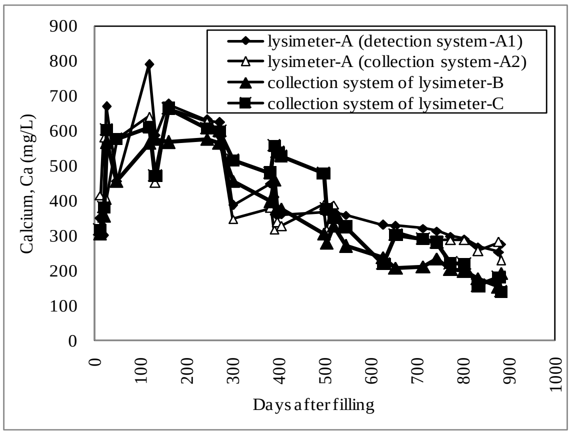

Figure 2 The changes of Ca concentration in leachate of open and sanitary landfill lysimeter

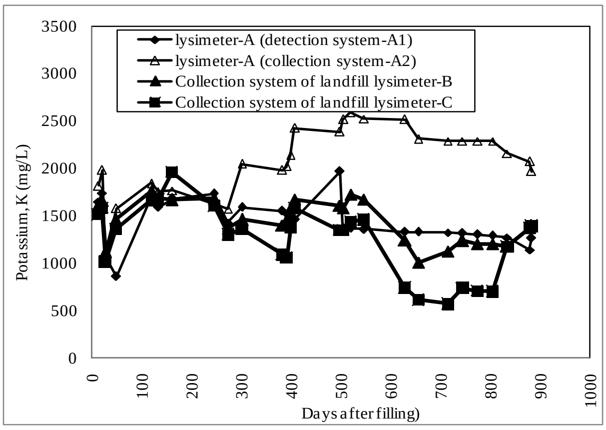

Figure 3 The changes of K concentration in leachate of open and sanitary landfill lysimeter

Potassium and Sodium Concentrations

The concentration of potassium is immobilized in the easily degradable organic matter, but it becomes soluble due to the decomposition processes within the MSW landfill (Burton et al. 2001). Potassium ion has the least complex forming ability and gets associated to the other ions that can be leached at first from the MSW bed (Brady and Weil, 1999; Yoshida et al. 2002).

The Figure 3 reveals the variation of K concentration with in relation to the increasing of elapsed period in case of leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A and sanitary landfill lysimeter-B and C. The Figure 3 depicts that the K concentration was ranging of 855.9 to 1967.0 mg/L, 1102.1 to 2589.0 mg/L, 1008.6 to 1764.3 mg/L and 567.2 to 1956.7 mg/L, for leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter-B and C, respectively. At the beginning of open dump lysimeter operation, K concentration was determined as 1650.5 and 1815.8 mg/L and after 50 day, it was dropped markedly to 855.9 and 1576.8 mg/L, and at 500 day, it was reached as maximum values of 1370 and 2590 mg/L, for leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, respectively. However, at the beginning of the sanitary landfill lysimeter operation, K concentration was measured as 1605.9 and 1509.9 mg/L for sanitary landfill lysimeter--B and C, respectively. There was no considerable change of K concentration for both the leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A after 600 day of operation.

Here, it is interesting to note that the highest concentration of K was measured in the leachate collection (A_2) system with 2589.0 mg/L, whereas, the lowest of 567.2 mg/L in sanitary landfill lysimeter-C. Here, it can be established that the variation of K concentration may be occurred regarding to entire lysimeter operation system due to provide of 400 mm thick CCL as a barrier between leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, simulation behaviour of open dumping and base liner as well as sanitary landfill and cap liner in case of open dump lysimeter-A and sanitary landfill lysimeter-B as well as the difference of sanitary landfill lysimeter-B and C in terms of thickness and compaction conditions of cap liner.

The variation of Na concentration with in relation to the increasing of elapsed period in case of leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A and sanitary landfill lysimeter-B and C is evident in Figure 4. Based on Figure 4 it can be depicted that Na concentration was ranging of 1267.9 to 2424. mg/L, 1251.6 to 2369.6mg/L, 1256.3 to 2764.3 mg/L and 1011.9 to 2409.1 mg/L, for leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter--B and C, respectively. Here, it is interesting to note that highest concentration of Na was measured in case of sanitary landfill lysimeter-B with 2764.3 mg/L, whereas, the lowest of 1011.9 mg/L in sanitary landfill lysimeter-C and the concentration of K was differed regarding to entire lysimeter operation systems.

Here, it is interesting to note that the trend for both the concentrations of K and Na were found as almost similar. A study conducted by Bilgili et al. (2006) stated that the Na and K concentrations of leachate were in the same trend with chloride Cl^- concentration. This situation was confirmed by Kimmel and Braids (1980) and also Sinan et al. (2006) who showed a linear relationship between Na and Cl^- concentrations in water samples obtained from a leachate plume at different distances from a landfill. So, finally it can be concluded that the finding in terms of Cl^-, K and Na concentrations in the present study were agreed well with the postulation given by Kimmel and Braids (1980) and Sinan et al. (2006). Based on the results of K and Na in terms of minimum and maximum concentrations with respect to entire lysimeter operation systems as shown in Table 1, these results are in agreement with the results obtained by Sinan et al. (2006) and also Diaz et al. (1996). In contrast, the minimum concentrations of both the K and Na exceed the minimum limit stated by Diaz et al. (1996).

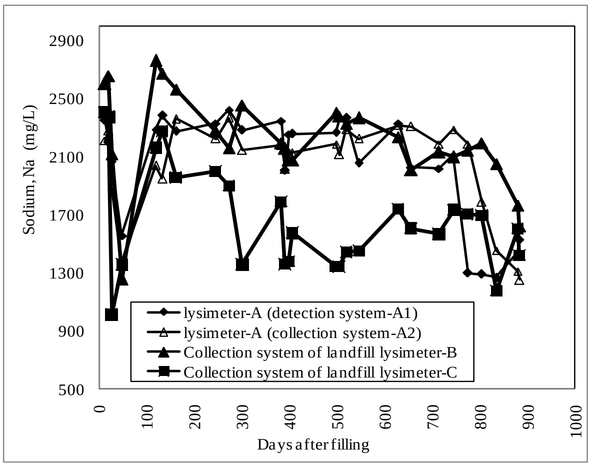

Figure 4 The changes of Na concentration in leachate of open and sanitary landfill lysimeter

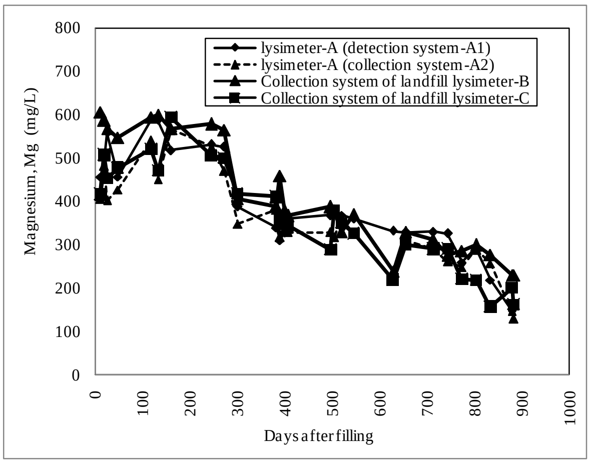

Figure 5 The changes of Mg concentration in leachate of open and sanitary landfill lysimeter

Magnesium Concentration

The Figure 5 reveals the variation of Mg concentration with in relation to the increasing of elapsed period in case of leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A and sanitary landfill lysimeter-B and C. The Figure 5 depicts that the Mg concentration was ranging of 147.25 to 589.9 mg/L, 127.15 to 564.2 mg/L, 228.4 to 605.9 mg/L and 156 to 594.2mg/L, for leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter-B and C, respectively. At the beginning of the open dump landfill lysimeter operation, Mg concentration was determined as 455.7 and 405.5 mg/L and at 120 day operation, it was reached as maximum values of 589.9 and 540.6 mg/L for leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, respectively. However, at the beginning of the sanitary landfill lysimeter-operation, Mg concentration in leachate was measured as 605.9 and 415.8 mg/L for sanitary landfill lysimeter-B and C, respectively. Here it can be concluded that the concentration of Mg was decreased markedly until the end of this trial, there was no considerable change in both the leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A after 650 day of operations.

Here, it is interesting to note that the highest concentration of Mg was measured in sanitary landfill lysimeter-B with 605.9 mg/L, whereas, the lowest of 127.15 mg/L was in the collection system open dump lysimeter-A. Here, it can be established that the variation of K concentration may be occurred with regarding to entire lysimeter operation system due to provide of 400 mm thick CCL as a barrier between leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, simulation behaviour of open dumping and base liner as well as sanitary landfill and cap liner in case of open dump lysimeter-A and sanitary landfill lysimeter-B as well as the difference of sanitary landfill lysimeter-B and C in terms of thickness and compaction conditions of cap liner.

Heavy Metal Concentrations

The main processes for the low metal concentrations in leachate are sorption and precipitation. Solid wastes contain soils and organic matter, which have a significant sorptive capacity, especially at neutral to high pH prevailing in methonegenic leachate (Bozkurt et al. 1999). However, solubility's of the metals with sulfides and carbonates is low and sulfide precipitation is often cited as an explanation for low concentrations of heavy metal in leachate. Sulfides and carbonates are capable of forming precipitates with heavy metals of Cd, Ni, Zn, Cu and Pb. In general, sulfide precipitation is expected to dominate heavy metal attenuation as compared to complexation agents (Reinhart et al. 1998). Cr is an exception to this because it does not form an insoluble sulfide precipitate. It tends to form insoluble precipitates with hydroxide (Revans et al. 1999 and Kjeldsen et al. 2002).

Occasionally, phosphate and hydroxides will also precipitate metals and hydroxide precipitates forms at pH at or above neutral, which is typically the case in methanogenic leachate (Reinhart et al. 1998). Other factors that will influence metals solubility include the cation exchange capacity of the refuse and how it changes during refuse oxidation and the presence of more oxidixed functional groups on the solid humic matter as well as humic matter in leachate (Martensson et al. 1999). Heavy metal leachability from the fresh waste is mainly depends on complexation and sorption within the lysimeter. In mined waste lysimeter, the major mechanism like precipitation and cation exchange capacity are accounts for maximum heavy metal content in leachate. The concentration of Cd, Cu, Cr, Mn, Ni, Zn and Pb in leachate in case of leachate detection (A_1) and leachate collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter-B and C is evident in Table 2 and also in Figures 6-13 and hence discussed in followings.

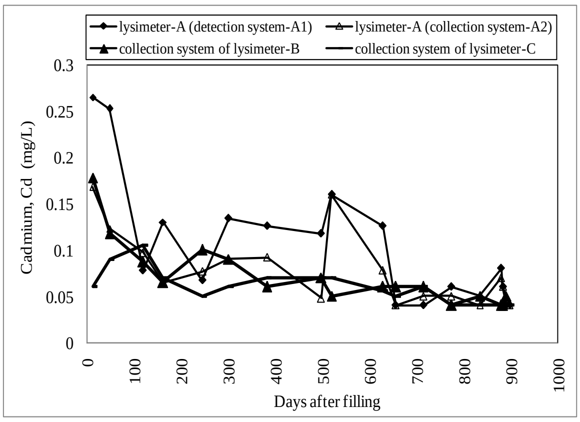

Figure 6 The changes of Cd concentrations in leachate of open and sanitary landfill lysimeter

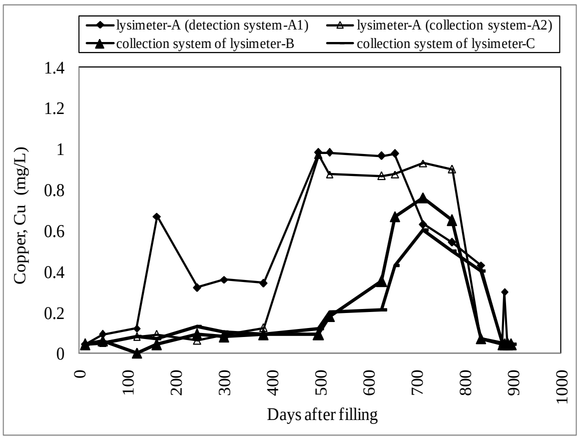

Figure 7 The changes ofCu concentrations in leachate of open and sanitary landfill lysimeter

Cd concentration was measured in the range of 0.04 to 0.265, 0.04 to 0.168, 0.04 to 0.178 and 0.04 to 0.105, while, Cu was ranging of 0.04 to 0.98, 0.04 to 0.97, 0.04 to 0.76 and 0.04 to 0.60 with in relation to the changing of elapsed period from waste deposition, for leachate detection (A_1), leachate collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter-B and C, respectively is evident in Figure 6. At the beginning of open dump lysimeter operation, the Cd concentration was found 0.265 and 0.168 mg/L, and after 120 day, it was decreased markedly 0.078 and 0.097 mg/L, for leachate detection (A_1), leachate collection system (A_2), respectively. At the day of 660 to until the end of this trial, there was no significance change for both the cases of open dump lysimeter. Moreover, at the beginning of open dump lysimeter operation, the Cu concentration was found 0.04 and 0.04 mg/L, and after 100 day operation, it was increased rapidly up the day of operation 500, 0.98 and 0.97 mg/L, for the leachate detection (A_1), leachate collection system (A_2), respectively (Figure 7). At the day of 630 and 775 to until the end of this trial, it was decreased rapidly for both the cases of open dump lysimeter system. In the other side, both the Cd and Cu concentration in case of sanitary landfill lysimeter-B and C were found the almost similar trend of open dump lysimeter operation, although the values of Cd and Cu in case of open dump lysimeter operation were found as higher than the case of sanitary landfill lysimeter-operation. Finally, it can be concluded that the findings in the present study in case of Cd and Cu concentration exceed the lower limit reported by Jensen et al. (1999); Kjeldsen et al. (2001); Kalyuzhnyi et al. (2004); Sinan et al. (2006); WasteSafe 2010 and Diaz et al. (1996).

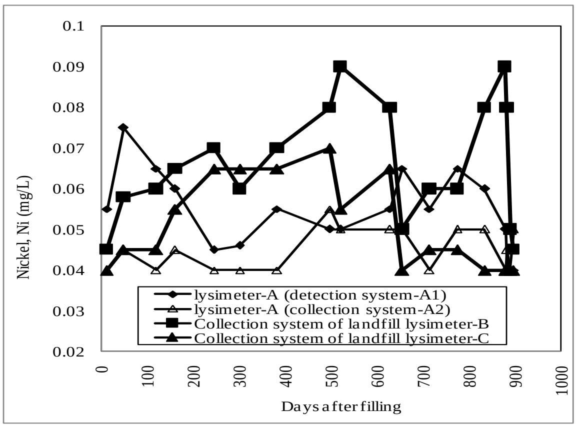

Figure 8 The changes of Ni in leachate of open and sanitary landfill lysimeter

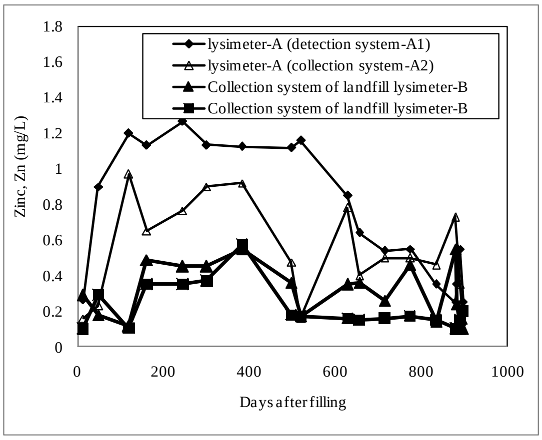

Figure 9 The changes of Zn in leachate of open and sanitary landfill lysimeter

The Figures 8 and 9 illustrates the variation of Ni and Zn concentration with in relation to the increasing of elapsed period from waste deposition for both the operation of open dump and sanitary landfill lysimeter. The Ni concentration was ranging of 0.04 to 0.075, 0.04 to 0.055, 0.04 to 0.09 and 0.04 to 0.07, while, Zn was ranging of 0.25 to 1.267, 0.15 to 0.97, 0.10 to 0.55 and 0.10 to 0.576, with in relation to the changing of elapsed period, for leachate detection (A_1), leachate collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter-B and C, respectively. From the beginning of sanitary landfill lysimeter-operation to up the day of 520, the Ni concentration was increased markedly and reached to the values of 0.09 and 0.07 mg/L, and after 520 day operation, it was dropped, for lysimeter-B and C, respectively. Moreover, from the beginning of sanitary landfill lysimeter-operation to up the day of 380, the Zn concentration was increased rapidly with 0.55 and 0.576 mg/L, for lysimeter-B and C, respectively.

In the other side, both the Ni and Zn concentration in case of sanitary landfill lysimeter-B and C were found as almost similar trend of open dump lysimeter operation, although the values of Ni and Zn in case of sanitary landfill lysimeter-operation were found as higher than that of open dump lysimeter operation. However, a study conducted by Sinan et al. (2006) reported that the Ni and Zn concentration were found as higher in case of sanitary landfillreactors than that of open dump reactors. So, it can be concluded that the findings in terms of Ni and Zn concentrations in the present study were agreed well with the statement stated by Sinan et al. (2006).

The variation of Cr and Mn concentration with in relation to the increasing of elapsed period are shown in Figure 10. At the beginning of open dump lysimeter operation, Cr concentration was found 0.065 and 0.06 mg/L, for leachate detection (A_1) and leachate collection system (A_2), respectively. At the day of 120, Cr concentration was decreased as 0.02 and 0.02 mg/L for both the cases, and after 120 day to end of this trail, the leachate detection system showing the higher than the leachate collection system. Here, it can also be noted that the Cr concentration for sanitary landfill lysimeter-operation were found as higher than that of open dump system, however, the concentration of Cr was found as higher in case of sanitary landfill lysimeter-B, with 0.09 mg/L. Moreover, a study conducted by Sinan et al. (2006) reported that the concentration of Cr was found as higher in case of sanitary landfillreactors than that of open dump reactors. So, it can be concluded that the findings in the present study were agreed well with the statement stated by Sinan et al. (2006).

In addition, the Mn concentration was ranging of 0.80 to 6.0, 2.5 to 21.2, 0.50 to 19.3 and 0.80 to 8.0 mg/L, with in relation to the changing of elapsed period, for the leachate detection (A_1), leachate collection system (A_2, sanitary landfill lysimeter-B and C, respectively (Figure 11). However, the highest concentration of Mn was found for the collection system of open dump lysimeter-A with 21.2 mg/L and both the open dump and sanitary landfill lysimeter-operation showing almost the same concentration of Mn.

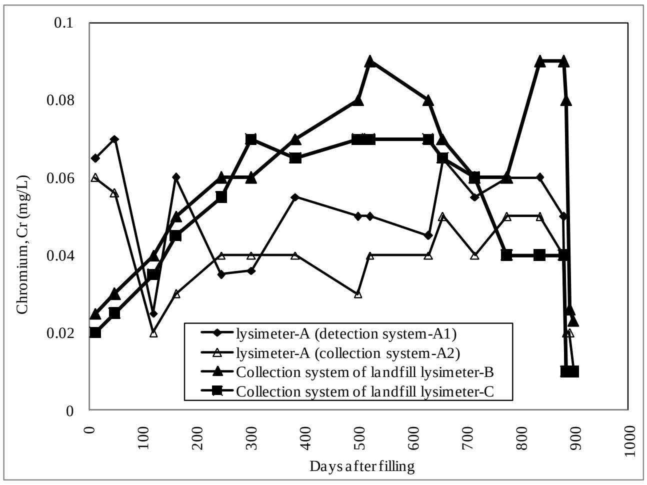

Figure 10 The changes of Ni in leachate of open and sanitary landfill lysimeter

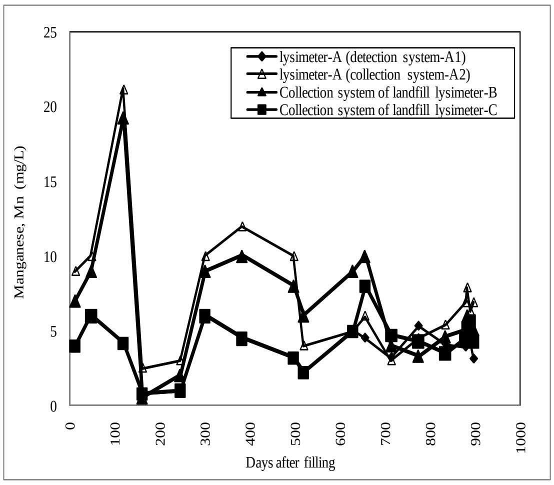

Figure 11 The changes of Ni in leachate of open and sanitary landfill lysimeter

The Figure 12 illustrates the variation of Pb concentration with in relation to the increasing of elapsed period for both the operation of open dump and sanitary landfilll lysimeter. The Pb concentration was ranging of 0.12 to 0.32, 0.12 to 0.397, 0.25 to 0.78 and 0.10 to 0.476 mg/L, with in relation to the changing of elapsed period, for the leachate detection (A_1), leachate collection system (A_2) of open dump lysimeter-A, sanitary landfill lysimeter-B and C, respectively. Here, it is interesting to note that the highest concentration of Pb was measured in sanitary landfill lysimeter-B with 0.78 mg/L, whereas, the lowest of 0.12 mg/L in both the cases of leachate detection system of open dump lysimeter-A and the sanitary landfill lysimeter-C and the concentration of Pb was differed regarding to entire lysimeter operation system.

Here, it can be established that the variation of Pb concentration may be occurred with regarding to entire lysimeter operation system due to provide of 400 mm thick CCL as a barrier between leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, simulation behaviour of open dumping and base liner as well as sanitary landfill and cap liner in case of open dump lysimeter-A and sanitary landfill lysimeter-B as well as the difference of sanitary landfill lysimeter-B and C in terms of thickness and compaction conditions of cap liner.

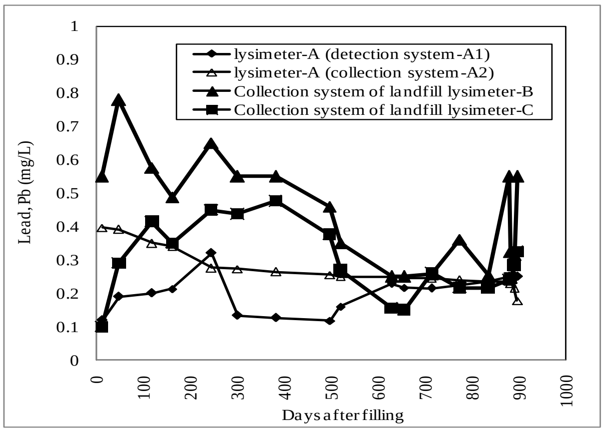

Figure 12 The changes of Ni in leachate of open and sanitary landfill lysimeter

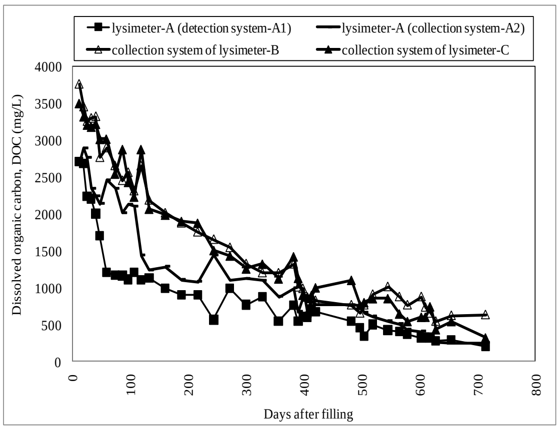

Figure 13 Leaching behavior of DOC in leachate of lysimeter at varying operational condition

Behavior of DOC

DOC values in leachate of sanitary lysimeter-B were relatively high (up to 3760 mg/l), and rapid reduction in DOC levels in A_1 and A_2 system of open dump lysimeter-A compared to collection system of sanitary lysimeter-B and C clearly showed the acceleration of bio-stabilization in open dump lysimeter-A shown in **Figure 13**. However, the maximum concentrations of DOC (2707, 2890, 3760 and 3500 mg/L) were recorded within 20th day of lysimeter operation of A_1 and A_2 system of lysimeter-A as well as the collection system of lysimeter-B and C, respectively. In sanitary landfillcells, DOC decreased mainly due to dilution**.** However, it is very clear that DOC reduction in both the system of lysimeter-A is larger and caused by dilution, since initial concentrations were very close in both the sanitary and open dump lysimeter as well as pH, chloride and EC indicated rather higher dilution in sanitary lysimeter-B and C (data not shown). BOD_5 in leachate showed distinctly different pattern between three landfill lysimeter**.** Moreover, BOD_5 of A_2 system of lysimeter-A has decreased faster than DOC and became 450 mg/L after around day 900 (data not shown).

Behavior of Heavy Metal with DOC

It seems that DOC had more influence on heavy metal concentration leaching from lysimeter at varying operational condition can be seen from Figures 14-19. Heavy metal leaching, especially for Cu and Pb has been suggested to be strongly dominated by interaction with particulate matter and DOC. However, Figures 14 and 15 reveal approval correlation between DOC levels and Cu and Pb leaching was observed in this study. This indicates that leaching of these heavy metals was being controlled dominantly by DOC in lysimeter. Rather, it was controlled mainly by simple dissolution into water phase from the solid matrix at this early stage of landfilling.

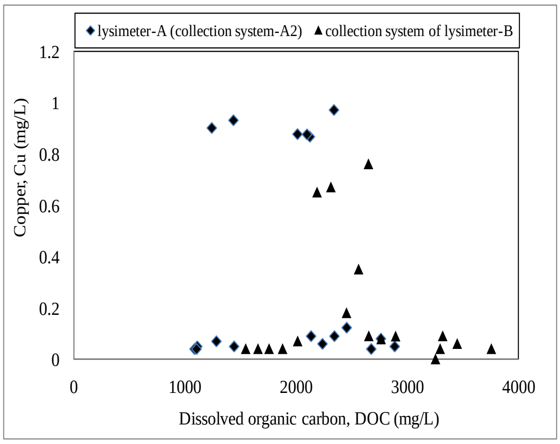

Figure 14 Relationship between Cu and DOC in leachate of lysimeter at varying operational condition

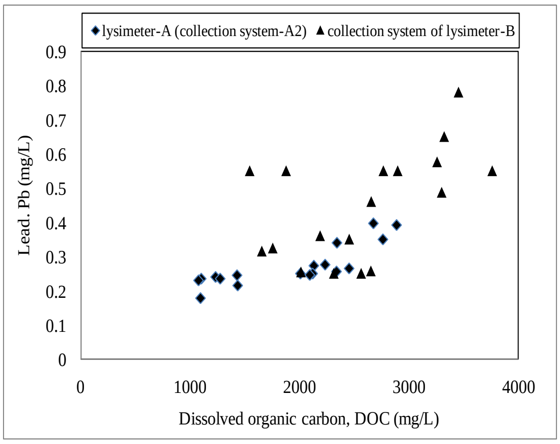

Figure 15 Relationship between Pb and DOC in leachate of lysimeter at varying operational condition

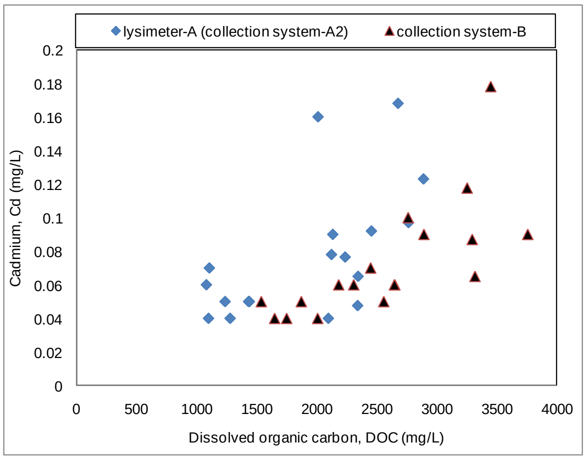

Figure 16 Relationship between Cd and DOC in leachate of lysimeter at varying operational condition

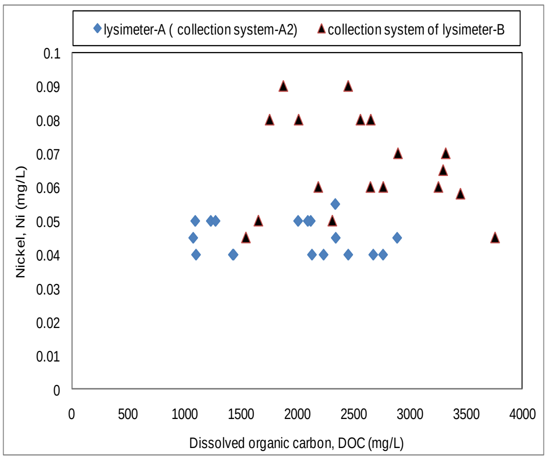

Figure 17 Relationship between Ni and DOC in leachate of lysimeter at varying operational condition

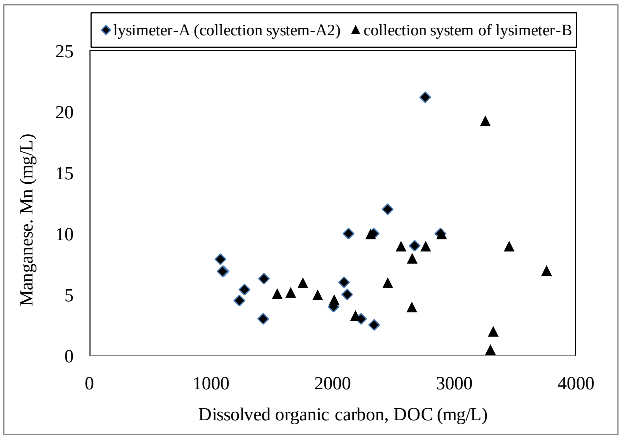

Figure 18 Relationship between Cu and DOC in leachate of lysimeter at varying operational condition

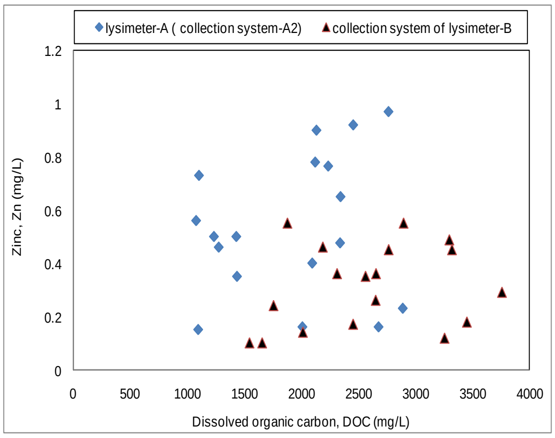

Figure 19 Relationship between Cu and DOC in leachate of lysimeter at varying operational condition

CONCLUSIONS

Result reveals that metal concentrations which were comparatively higher in leachate of open dump lysimeter were Ca and K, however, the heavy metal concentrations of Cd, Cu, Zn and Mn, and those apparently lower were metals of Na, Mg and Fe as well as heavy metals of Cr, Pb and Ni. Here, it can be established that the variation of metals concentration may be occurred regarding to entire lysimeter operation system due to provide of 400 mm thick CCL as a barrier between leachate detection (A_1) and collection system (A_2) of open dump lysimeter-A, simulation behaviour of open dumping and base liner as well as sanitary landfill and cap liner in case of open dump lysimeter-A and sanitary landfill lysimeter-B as well as the difference of sanitary landfill lysimeter-B and C in terms of thickness and compaction conditions of cap liner. There was no meaningful correlation between DOC and Cu and Pb for all the lysimeters. Further leaching tests and total content measurements on solid samples excavated from the test cells will provide further information on the effect of aerobic landfill bioreactor operation for leaching behavior of heavy metals.

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Formulation of Sub-Pollution Indices and Evaluation of Leachate Pollution Index of Pilot Scale Landfill Lysimeter

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ABSTRACT

The principal concern of this study is to formulate the sub-pollution indices (sub-LPIs) and evalaute the individual and overall leachate pollutant index (LPI) of pilot scale landfill lysimeter at KUET campus, Bangladesh. To these endeavors, leachate samples from collection chamber of landfill lysimeter were sampling and the relevant parameters required for evaluating LPI were measured and monitored in the laboratory. Both the open dump and sanitary landfill conditions having a base liner and two different types of cap liner were simulated. Three sub-LPIs in terms of LPI in organic pollutant (LPI_{or}), LPI in inorganic pollutant (LPI_{in}) and LPI in heavy metal (LPI_{hm}) as well as overall LPI had been developed and reported by the author. Here, it can be noted that component of organic fraction in leachate for entire lysimeter operating system had highest against the other counter fraction i.e. inorganic and heavy metal fraction and consequently shows the highest LPI_{or} than that of LPI_{in} and LPI_{hm}. Here, it can be depicted that sub-LPIs and overall LPI has decreased in relation to the increasing of elapsed period of MSW deposited in landfill lysimeter. Results showed that collection system of sanitary lysimeter-A had the highest sub-LPIs and overall LPI than that of other lysimeter operating system. Finally, it can be depicted that overall LPI was significantly higher for entire lysimeter operating system and proper treatment will be necessary before discharging the leachate into the water bodies.

Keywords: Landfill lysimeter, solid waste, aggregation function, sub-pollution indices, leachate pollution index.

INTRODUCTION

The term ‘landfill’ is used herein to describe a unit operation for final disposal of municipal solid waste (MSW) on land, designed and constructed with the objective of minimum impact to the environment (Rafizul and Alamgir 2012a). The term ‘landfill’ can be treated as synonymous to ‘sanitary landfill’ of MSW, only if the latter is designed on the principle of waste containment and is characterized by the presence of a liner and leachate collection system to prevent ground water contamination. Sanitary landfill is one of the secure and safe facilities for the disposal of MSW (Davis and Cornwell 1998). Moreover, it is a well-suited method for managing of MSW all over the world and to investigate the performance of sanitary landfill the behavioral patterns namely; leachate generation, landfill gas (LFG) emissions etc. are required (Visvanathan et al. 2002). Lysimeter is a simulate form of sanitary landfill in the sense of control device. The word lysimeter is a combination of two Greek words *“Lusis” means “Solution”* and *“Metron” means “Measure”* and the original aim is to measure soil leaching (Rafizul et al. 2012b). As a result of the serious environmental problems associated with abandoned dump sites and the high costs of clean-up measures to deal with the contaminated sites, almost all countries have introduced regulations to safeguard the water aquifers from the leachate generated from the landfills (Rafizul et al. 2011).

Leachate is characterized by its high content of organic constituents, metals, acids, dissolved salts and microorganisms (Orta et al. 2003). Containing hundreds of different chemicals, the characteristics of leachate vary significantly with respect to its composition, volume, and the presence of biodegradable matter and with time (Chu et al. 1994). Leachate constitutes a flow that is highly aggressive and dangerous to the environment, with a contamination potential exceeding that of several industrial-waste materials. The indices allows for the representation of a complex set of information about ecosystem variables in a simple fashion (Zandbergen and Hall, 1998). A technique to quantify the leachate contamination potential of landfills on a comparative scale by using an index known *LPI* has been developed and reported elsewhere (Kumar and Alappat 2003). LPI provides an overview of leachate contamination potential of a landfill on a comparative scale. In an effort to effectively communicate the dominating pollutants present in leachate sample, it was decided to subgroup the pollutants considered in the LPI. The formulation and applications of the sub-indices of the LPI (sub-LPIs) and the overall LPI using leachate characteristics of landfill lysimeter at KUET is presented in this paper.

OVERVIEW OF LANDFILL LYSIMETER

Three landfill lysimeters designated as A, B and C were designed and hence constructed at KUET campus, Bangladesh. The operational condition, liner specifications, simulation behaviour and the total weight of MSW deposited in each lysimeter can be obtained in Rafizul et al. [1] and presents in Table 1. The MSW deposited in each lysimeter mainly consists of 93 (w/w) organic (food and vegetables), 3 (w/w) of plastic/polythene and 2 (w/w) of leather/rubber, 1 (w/w) of animal bone and rubber/leather as well as 1 (w/w) of rope/straw and egg pill. However, the organic and moisture content of MSW was found 52 and 65%, respectively, and the total volume was 2.80 m^3 (height 1.6 m) with a manual compaction to achieve the unit weight of 1,064 kg/m^3. At the bottom of each lysimeter, a concrete layer of 125 mm thickness was provided then the lysimeter was filled with stone chips (diameter 5-20 mm) and coarse sand (diameter 0.05- 0.4 mm) to the height of 15cm of each to ensure uniform and uninterrupted drainage. At the base of each lysimeter after placing the perforated leachate collection pipe, a geo-textile sheet having 0.60 m wide and 1.65 m length was placed to avoid a rapid clogging by the sediments.

Table 1 Operational conditions of lysimeter to simulate different landfill conditions

lysimeter	*Operating condition*	*Refuse (kg)*	*Liner specification*	*Simulation*
A	Open dump lysimeter with leachate detection (A_1) system Open dump lysimeter with leachate collection (A_2) system	2860	400mm thick CCL as a barrier between leachate detection and collection system of lysimeter-A	present practice of open dumping
B	Sanitary landfill lysimeter with gas collection and leachate recirculation system	2985	Cap liner-I (300mm thick CCL)	applicability of designed top cover
C		2800	Cap liner-II (900mm thick natural top soil)	

Landfill lysimeter-A (Open Dump)

In open dump lysimeter-A, a compacted clay liner (CCL) of 400 mm thickness was placed as the base liner and a layer of compost of 150 mm thick was used as the top cover to simulate the behaviour of present practice of open dumping in Bangladesh shown in **Figure 1.** This lysimeter was operated at two operational condition in terms of leachate detection (A_1) and collection (A_2) system. In lysimeter-A, MSW was not covered by a top cover to pervert the movement of air, water and LFG. Moreover, the thickness of MSW is such that it is expected the atmospheric air can move in the entire MSW deposited in this cell with negligible inference. Due to the mentioned practical situations, lysimeter-A represents an open dump condition.

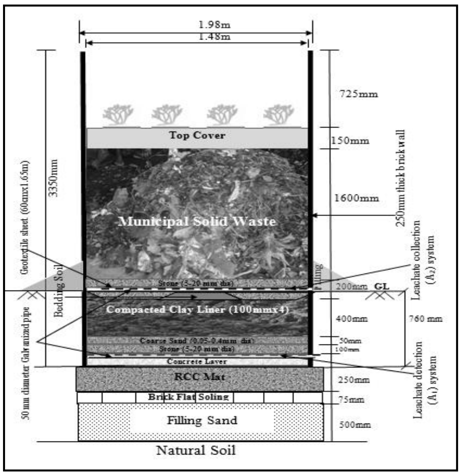

Figure 1 Schematic diagram of open dump lysimeter-A

Sanitary landfill lysimeter-B (Cap liner I)

In sanitary landfill lysimeter-B, the characteristics and volume of MSW was similar to that of the open dump lysimeter-A. However, it differs with open dump lysimeter-A, by a top cover and without a base liner, because this cell aims to examine the applicability of the designed top cover to simulate the sanitary landfill condition. The top cover consists of stone chips (diameter 5-20 mm) and coarse sand (diameter 0.05- 0.4 mm) layer each of 100 mm thickness, then a 300 mm CCL was provided. On the CCL, there were 150 mm thick coarse sand (diameter 0.05-0.4 mm) and 150 mm thick stone chips (diameter 5-20 mm), which was followed by 600 mm thick top soil shiwn in **Figure 2.** Due to the above mentioned practical situations, lysimeter-B represents a sanitary landfill condition and the flow rate and the composition of LFG is measure. In sanitary lysimeter-B, 38 mm diameter of gas collection and 25mm diameter of leachate recirculation pipe were installed.

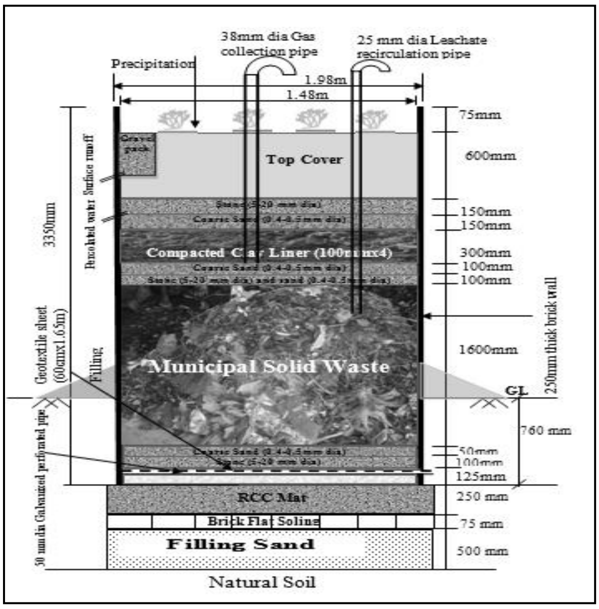

Figure 2 Schematic diagram of sanitary landfill lysimeter-B

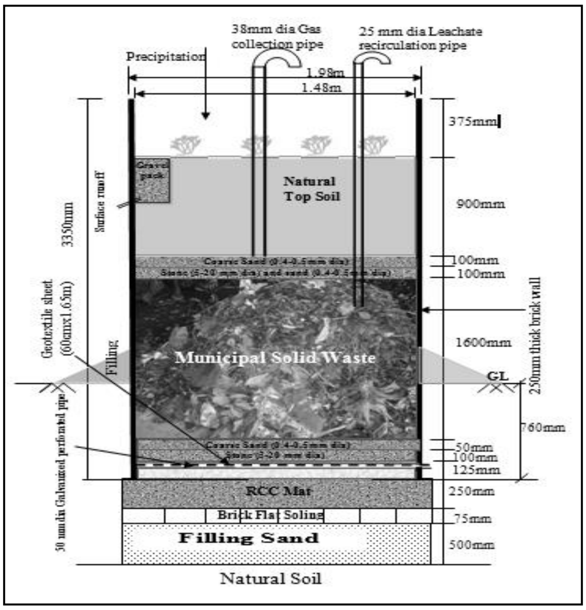

Figure 3 Schematic diagram of sanitary landfill lysimeter-C

Sanitary landfill lysimeter-C (Cap liner II)

In sanitary landfill lysimeter-C, there was also no base liner and the provided top cover was different than that of sanitary landfill lysimeter-B. In this lysimeter no CCL was used; however, 900 mm thick natural topsoil was used instead of 300 mm thick CCL and 600 mm thick top soil shown in **Figure 3.** Moreover, the drainage and gas collection layers were remained same as the sanitary landfill lysimeter-B. Designated compaction of the CCL in the lysimeter means the degree of compaction which was provided in the pilot scale sanitary landfill (PSSL) at Rajbandh, Khulna. To achieve the designated compaction at the CCL of lysimeter, locally manufactured hammer similar to that used in the PSSL was employed.

MATERIALS AND METHODOLOGY

In this study, to derive sub-leachate pollution indices as well as individual and overall pollutant rating of landfill lysimeter, the detailed procedure advocated by Kumar and Alappat [9] was followed and hence discussed in followings.

Concept of leachate pollution index

The LPI represents the level of leachate contamination potential of a given landfill. It is a single number ranging from 5 to 100 (like a grade) that expresses the overall leachate contamination potential of a landfill based on several leachate pollution parameters at a given time.

Variables weight facors and curves

The weight factor indicates the importance of each pollutant variable to the overall LPI provided in Table 1.The averaged sub-index curves for all the pollutant variables have been reported by Kumar and Alappat (2003).

Variable aggregation function

The weighted sum linear aggregation function was used by Kumar and Alappat (2003) to sum up the behavior of all the leachate pollutant variables. However, Panelists suggested that if the concentrations of the *eighteen (18)* selected variables are known, the following *Equation (1)* is used. Otherwise, *Equation (2)* is used.

$$LPI = \sum_{i=1}^{n} w_i p_i \tag{1}$$

LPI = the weighted additive leachate pollution index, w_i = the weight for the I^{th} pollutant variable, p_i = the sub-index value of the I^{th} leachate pollutant variable, number of leachate pollutant parameters, n =18 and $\sum w_i = 1$.

$$LPI = \frac{\sum_{i=1}^{m} w_i p_i}{\sum_{i=1}^{m} w_i} \tag{2}$$

Pollutant parameter for which data is available in this study, m < 18 and $\sum_{i=1}^{m} w_i < 1$

Table 2 Weight factors of leachate parameters based on *sub-LPI (after Kumar and Alappat (2003))*

Index	Parameters	Weight factor
$LPI_{organic}$ (LPI_{or})	Chemical oxygen demand (COD)	0.267
	Biological oxygen demand (BOD_5)	0.263
	Phenol compound	0.246
	Total coliform bacteria (TCB)	0.224
	Summation	1.000
$LPI_{inorganic}$ (LPI_{in})	pH	0.214
	Total kjeldahl nitrogen (TKN)	0.2060
	Ammonia nitrogen (NH_4-N)	0.198
	Total dissolved solid (TDS)	0.195
	Chloride (Cl^-)	0.187
	Summation	1.000
$LPI_{heavy\ metal}$ (LPI_{hm})	Total chromium (Cr)	0.125
	Lead (Pb)	0.123
	Mercury (Hg)	0.121
	Arsenic	0.119
	Cyanide	0.114
	Zinc (Zn)	0.11
	Nickel (Ni)	0.102
	Copper (Cu)	0.098
	Total iron (Fe)	0.088
	Summation	1.000

Sub-indices of leachate pollution index

To make LPI is more informative and useful among the scientific community and field professionals, the LPI may be subdivided into three sub-indices in terms of LPI in organic pollutant (LPI_{or}), LPI in inorganic pollutant (LPI_{in}) and LPI in heavy metal (LPI_{hm}) provided in Table 2. In the group of LPI_{or}, organic compounds are normally composed of a combination of carbon, hydrogen and oxygen, with nitrogen in some cases. In this group, pollutants selected for sub-LPI are BOD_5, COD, phenol compounds and TCB. The weight factors for the pollutants in *LPIor* have been recalculated on a scale of 1 (Table 2). In contrary, chlorides, alkalinity, various forms of nitrogen, phosphorous, sulphur, pH, heavy metals, gases like hydrogen sulphide and methane, etc. constitute the inorganic component of leachate. The LPI inorganic component consists of Cl^-, pH, NH_4-N, TKN and TDS. The weight factors for the pollutants in LPI_{in} have been recalculated on a scale of 1 as if LPI_{in} is an absolute index. Moreover, many metals such as Cr, Pb, Zn, Ni, Cu, Fe and Hg are important constituents of leachates produced from landfills and considered in LPI_{hm} group. Two non-metal pollutants, arsenic and cyanide have also been included in this sub-group. The weight factors for the pollutants have been recalculated on a scale of 1.

Table 3 Characteristics of leachate, sub-*LPI* and overall *LPI* landfill lysimeter at elapsed period 7 day

Index (1)	Parameter (2)	Weight factor, w_i (3)	Pollutant concentration, c_i (4)				Individual pollutant rating, p_i (5)				Overall pollutant rating, w_ip_i (6)			
			A_1	A_2	B	C	A_1	A_2	B	C	A_1	A_2	B	C
LPI_{or}	COD	0.267	22650	60000	60000	56490	84	94	94	93	22.43	25.10	25.10	24.83
	BOD_5	0.263	2080	2860	2790	2286	41	46	45	42	10.78	12.10	11.84	11.05
	Phenol	-	-	-	-	-	-	-	-	-	-	-	-	-
	TCB	0.224	6540	8280	8200	8230	85	89	90	90	19.04	19.94	20.16	20.16
	Total	**0.754**									52.25	57.13	57.09	56.04
LPI											**69.30**	**75.77**	**75.72**	**74.32**
LPI_{in}	pH	0.214	6.87	7.87	7.92	7.38	6.0	5.0	5.0	6.0	1.28	1.07	1.07	1.28
	TKN	0.206	1010	2180	1430	1340	33	78	50	45	6.80	16.07	10.30	9.27
	NH_4-N	0.198	705	897	997	920	77	95	99	97	15.25	18.81	19.60	19.21
	TDS	0.195	9876	35670	29120	26580	21	83	69	63	4.10	16.19	13.46	12.29
	Cl^-	0.187	3037	3572	1350	760	23	31	12	8	4.30	5.80	2.24	1.50
	Total	**1.000**									31.72	57.93	46.67	43.54
LPI											**31.72**	**57.93**	**46.67**	**43.54**
LPI_{hm}	Cr	0.125	0.08	0.21	0.17	0.09	5.0	5.0	5.0	5.0	0.63	0.63	0.63	0.63
	Pb	0.123	0.41	0.55	0.92	0.65	7.0	7.0	9.0	8.0	0.86	0.86	1.11	0.98
	Hg	-	-	-	-	-	-	-	-	-	-	-	-	-
	As	0.119	0.01	0.04	0.03	0.02	5.0	5.0	5.0	5.0	0.60	0.60	0.60	0.60
	Cn	-	-	-	-	-	-	-	-	-	-	-	-	-
	Zn	0.11	1.40	1.50	0.98	0.65	5.5	5.0	5.0	5.0	0.61	0.55	0.55	0.55
	Ni	0.102	0.10	0.19	0.13	0.12	5.0	5.0	5.0	5.0	0.51	0.51	0.51	0.51
	Cu	0.098	1.30	1.60	1.10	1.20	7.0	7.0	7.0	7.0	0.69	0.69	0.69	0.69
	Fe	0.088	25.90	45.70	43.50	38.70	5.5	6.0	6.0	6.0	0.48	0.53	0.53	0.53
	Total	**0.765**									4.37	4.36	4.60	4
LPI											**5.71**	**5.69**	**6.01**	**5.85**
	LPI	$(0.175\mathbf{LPI_{or}}+0.257\mathbf{LPI_{in}}+0.391\mathbf{LPI_{hm}})/0.823$									***27.35***	***36.91***	***33.53***	***32.18***

All values in mg/L except pH and total coliform unit (cfu/100ml)

Calculation of sub-pollution indices and overall LPI: Case study

For calculating sub-pollution indices (sub-LPI) and overall LPI, the following steps were followed:

1. The concentration of fifteen (15) leachate parameters for *sub-LPI* and overall *LPI* were measured and monitored in the laboratory at a regular interval of time up to the elapsed period 900 days. Moreover, the concentration of leachate at elapsed period 7 days is provided in *Table 3, column 4*.
2. Then evaluating sub-index scores (*pi*) of all the pollutants included in *sub-LPI* and overall *LPI* based on variables rating curves with respect to their concentration (*Table 3, column 5*).
3. The sub-pollution indices in terms of $\boldsymbol{LPI}_{or}$, $\boldsymbol{LPI}_{in}$ and $\boldsymbol{LPI}_{hm}$ are calculated using the weight factors given in Table 2 based on the aggregation function from Equation (2) (*Table 3, column 6*).
4. Finally, the aggregation of the three sub-*LPIs* gets the overall *LPI*. The three sub-*LPI* values are aggregated to calculate the overall *LPI* using following Equation (3).

$$\boldsymbol{LPI} = (0.175\boldsymbol{LPI}_{or}+0.257\boldsymbol{LPI}_{in}+0.391\boldsymbol{LPI}_{hm})/0.823 \qquad (3)$$

Where *LPI* is the overall ***LPI***, $\boldsymbol{LPI}_{or}$ is the sub-leachate pollution index of organic component; $\boldsymbol{LPI}_{in}$ for inorganic component and $\boldsymbol{LPI}_{hm}$ for of heavy metal component in leachate. Here, it can be noted that Equation 3 has been derived based on the weight factors of pollutants included overall LPI and their contribution to each sub-*LPI*. However, the component of organic, inorganic and heavy metal is 17.5, 25.7 and 39.10 % for evaluating overall *LPI* and used to derive the Equation 3 .

RESULTS AND DISCUSSIONS

Table 3 illustrates the calculation of sub-LPI and overall *LPI* for a particular elapsed period of 7 days of leachate sampling. The detection (A_1) and collection (A_2) system of open dump lysimeter-A as well as the collection system of sanitary landfill lysimeter-B and C had the highest component of organic fraction against the other counter fraction i.e. inorganic and heavy metal fraction (Table 3). Consequently, all the lysimeter operating system shows the highest LPI_{or} than that of LPI_{in} *and* LPI_{hm} provided in **Figure 4.** Moreover, Table 2 reveals that organic fraction for A_2 system of lysimeter-A was highest and consequently the highest LPI_{or} than the other operating system provided in **Figure 5.**

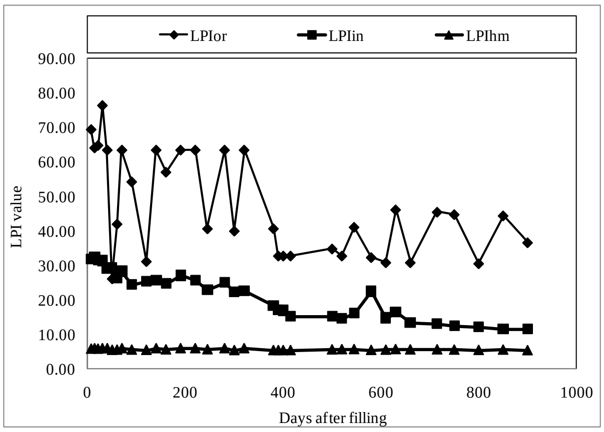

Figure 4 Sub-pollution indices of landfill lysimeter.

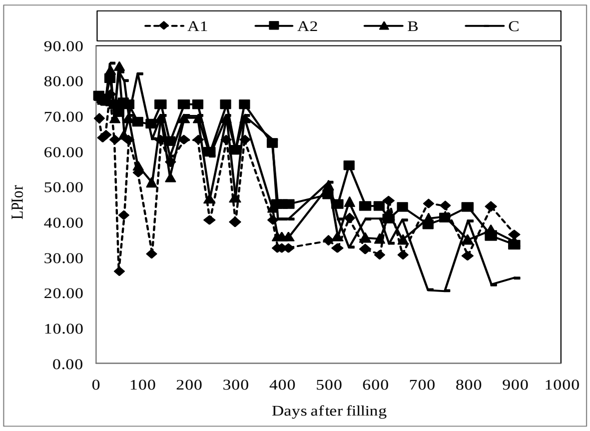

Figure 5 LPI in organic pollutant of open dump and sanitary landfill lysimeter.

Table 3 shows that values of BOD_5, TKN, TCB, TDS, chloride, Cr, As, Zn, Ni, Cu and Fe was highest and consequently the highest individual and overall LPI for collection (A_2) system of lysimeter-A. A significant difference between individual and cumulative pollution ratings for both the collection system of open dump lysimeter-A and sanitary lysimeter-C was observed due to the distinct difference in their concentrations (Table 3). All the concentrations except Cl^-, Zn and Cu were lower for the A_1 system of lysimeter-A than that of collection system of lysimeter-A, B and C and has lowest individual and cumulative pollution rating and consequently lower LPI shown in **Figure 6.** Here, it can be established that variation of leachate concentration in case of A_1 and A_2 system may be occurred due to the providing of 400 mm thick CCL as a barrier between the detection and collection system of lysimeter-

A. As the A_2 system of lysimeter-A was provided just below the MSW deposited in lysimeter-A and the followed A_1 was separated with the 400 mm thick CCL and this operational mode may be considered for the variation of leachate concentration. Moreover, the variation for the collection system of lysimeter-A and B may be occurred due to the simulation behaviour of open dumping and providing the base liner in open dump lysimeter-A as well as sanitary landfill and providing the cap liner in lysimeter-B. In contrary, the variation of leachate concentration (Table 3) for collection system of sanitary lysimeter-B and C may be occurred due to difference of lysimeter-B and C in terms of thickness and compaction conditions of cap liner.

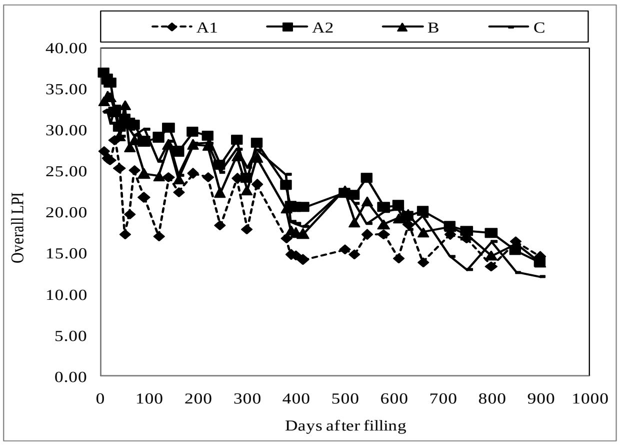

Figure 6 Overall LPI of open dump and sanitary landfill lysimeter.

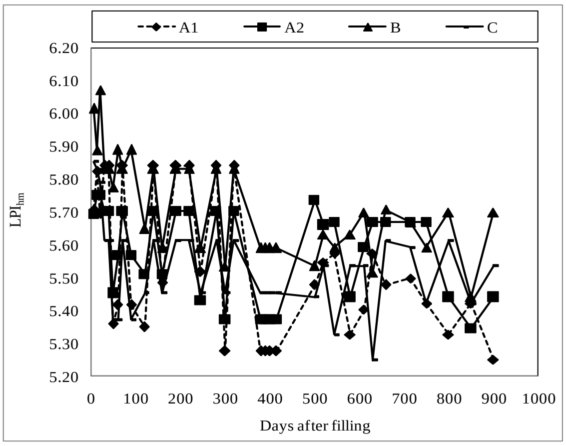

Figure 7 LPI in heavy metal of open dump and sanitary landfill lysimeter.

The LPI of A_2 system of lysimeter-A is slightly higher than the collection system of lysimeter-C but both these lysimeters have higher LPI than the collection system of lysimeter-B **(Figure 6).** This can be ascribed to the lower individual pollution ratings of A_1 system of lysimeter-A due to the relatively lower concentrations of all the pollutants except Cl^-, Zn and Cu. Here, it of interested to note that *LPI* has decreased in relation to the increasing of elapsed period provided in **Figure 6.** The overall LPI was higher for A_2 system of lysimeter-A because of the higher concentration of pollutant has the highest individual and cumulative pollution rating and consequently the higher LPI than the other operational system. Here, it is important to note that standards for the disposal of treated leachate to water as per the Management and Handling Rules (The Gazette of India 2000) should not exceed 2.0, 0.1, 250, 0.01, 30, 0.20, 0.20, 1.0, 5.0, 5.5-9.0, 100, 3.0, 50.0, 2100, 3.0 and1000 for Cr, Pb, COD, Hg, BOD_5, As, Cn, Phenol, Zn, pH, TKN, Ni, NH3-N, TDS, Cu and Cl^- concentration and their corresponding overall *LPI* of 7.38. The comparison of leachate characteristics with the standards for the disposal of treated leachate verifies the fact that the leachate generated from A_2 system of lysimeter-A is highly contaminated and *LPI* for entire lysimeter operating system exceed the *LPI* of treated leachate of 7.38. The high *LPI* demands that leachate generated from landfill lysimeter should be treated.

Moreover, it can be depicted that the comparatively lower LPI_{hm} for entire lysimeter operating system **(Figure 4)** are attributable to low concentrations of heavy metals in leachate (Table 2). Moreover, due to the lower concentration of heavy metal for the A_1 system of lysimeter-A, consequently shows the lowest LPI_{in} provided in **Figure 7.** In contrary, comparetively the higher inorganic compound than heavy metal implies the highest LPI_{in} than that of LPI_{hm} (Table 2). Moreover, comparetively the higher inorganic compounds implies the highest LPI_{in} for the A_2 system of lysimeter-A shown in **Figure 8.**

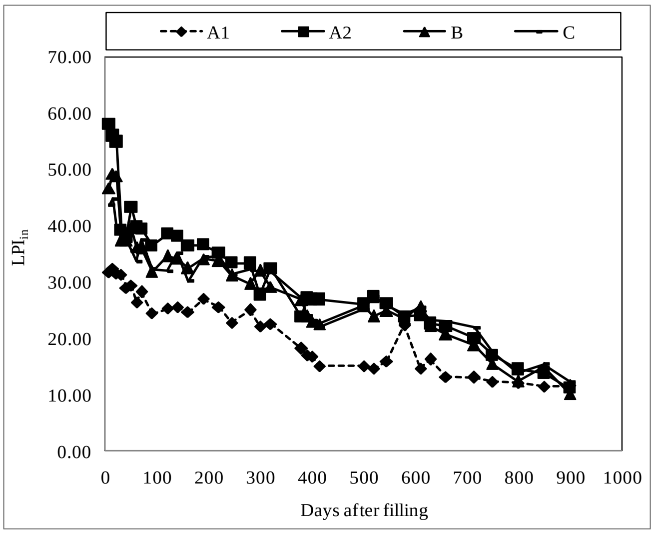

Figure 8 LPI in inorganic pollutant of open dump and sanitary landfill lysimeter.

CONCLUSIONS

The functionality of locally available construction materials used for solid waste landfill was judged in this study through the proper investigation and evaluation of leachate concentration, sub-pollution indices and overall LPI in three lysimeters from similar nature of MSW. The component of organic fraction in leachate for entire lysimeter operating system had highest against the other counter fraction i.e. inorganic and heavy metal fraction and consequently shows the highest LPI_{or} than that of LPI_{in} *and* LPI_{hm}. Here, it can be depicted that *LPI* has decreased in relation to the increasing of elapsed period of MSW deposited in lysimeter. Finally, it can be concluded that comparison of leachate characteristics with the standards set for the disposal of treated leachate verifies the fact that leachate generated from lysimeter is highly contaminated and *LPI* exceed the *LPI* of treated leachate and proper treatment to be ensured before discharging the leachate.

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Investigation of Solid Waste Management and Surrounding Groundwater Quality at Rajbandh Landfills Site

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ABSTRACT

Urban solid waste management is currently been regarded as one of the most immediate issues for city authorities according to the rapidly growing cities of developing countries. This paper proclaims the outline of a demonstration project that aims to develop a safe and sustainable system for the management of municipal solid waste at disposal site in Bangladesh through the practical application of improved sanitary landfill. The sanitary landfill of Rajbandh in Khulna is considered as the case study area. Groundwater samples were collected from one, two, and three km radius around the Rajbandh dumping site respectively. High concentrations of Chloride content (800 mg/L), Nitrate (3.2 mg/L) and Phosphate (1.2 mg/L) were noticed in the shallow groundwater within 1 km radius of the dumping site; which suggests the possible contamination of groundwater aquifer due to infringement of leachate beneath the landfills. Furthermore, a high concentration of organic matter was found in the collected water samples. Some other physicochemical parameters such as pH, Conductivity, Alkalinity, Hardness, Total Dissolved Solid (TDS), etc. were also investigated to depict a comprehensive scenario of the shallow groundwater aquifer around the disposal site. Finally, based on the major findings of this study, proposals were given for the sustainable development of the existing waste management practices at Rajbandh dumping site.

INTRODUCTION

Bangladesh is a densely populated country in the third world facing myriads due to the generation of huge volume of waste which is closely related with the growth of population. Due to intense population in urban areas, appropriate and safe solid waste management of municipal solid wastes (MSW) is of utmost importance to allow healthy living conditions for the population. MSW is considered as one of the major global environmental problems, especially in least developed Asian countries (LDACs) and most important solid waste because of its nature and impact on our community, which consists of hazardous and non-hazardous wastes (Zurbrugg, 2002). It is a fact that solid waste composition differs from one community to another community to their culture and socio-economic level. However, solving inadequate management of solid waste in general is very challenging because of its heterogeneous nature. On the other hand, solving the problem in urban area of developing countries is more challenging because of two factors: Low socio-economic level of the majority of population and their lack of awareness of scope of problem as well as lack of suitable of a suitable technology platform needed to face problem.

The oldest and most common way of disposing of solid wastes is open dump. Though in recent years thousands have been closed, many are still being used. In many cases, they are located wherever land is available, without regard to safety, health hazard and aesthetic degradation. The waste is often piled as high as equipment allows. In some instances, the refuse was ignited and allowed to burn. In others, the refuse was periodically leveled and compacted. As a general rule, open dumps tend to create a nuisance by being unsightly, breeding pests, creating a health hazard, polluting the air and sometimes polluting groundwater and surface water. Landfill is an engineered waste disposal site facility with specific pollution control technologies designed to minimize potential impacts. Landfills are usually either placed above ground or contained within quarries and pits

Landfills are sources of groundwater and soil pollution due to the production of leachate and its migration through refuse. Municipal solid waste did not pose a significant problem until human established settlements near landfill. Prior to that, the types and quantities of waste were readily degraded or consumed by animals or naturally degraded without causing significant impact to the environment and groundwater quality if leachate is discharging into these water bodies. Groundwater is that portion of subsurface water which occupies the part of the ground that is fully saturated and flows into a hole under pressure greater than atmospheric pressure. Groundwater occurs in geological formations known as aquifer. An aquifer (gravel/ sand) may be defined as a geologic formation that contains sufficient permeable materials to yield significant quantities of water to wells and springs. This implies the ability of the formation to store and transmit water. Groundwater is an important source of drinking water for humankind. It contains over 90% of the fresh water resources and is an important reserve of good quality water. This study aims at investigating the waste management practices and consequent groundwater contamination around Rajbandh dumping site and hence proposing guidelines for modification.

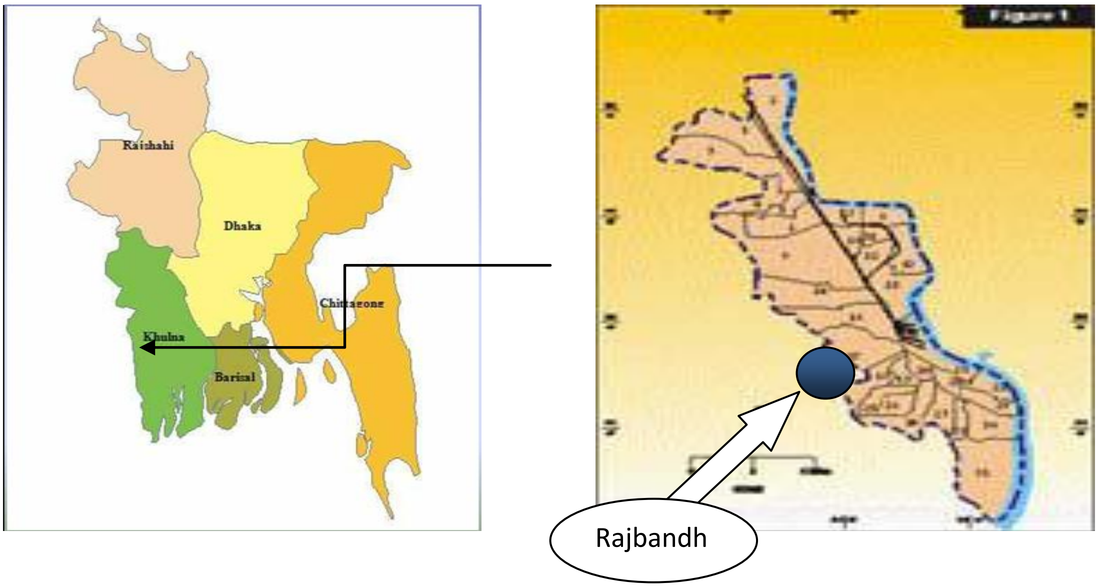
OVERVIEW OF STUDY AREA

Selection of Study Area

Khulna, the third largest metropolitan city of Bangladesh, stands on the banks of the Rupsha and the Bhairab rivers. It is in the south-western part of the country with its location on the axis of Jessore-Mongla port. Geographically, Khulna lies between 22047´16´´to 22052´ north latitude and 89031´36´´ to 89034´35´´ east longitude. The city is 4 m above the mean sea level (MSL). At present, Khulna city has a population of about 1.5 million with an area of 47 square kilometers and 31 Wards.

Khulna city is located on natural levees of the Rupsha and Bhairab rivers and characterized by Ganges tidal floodplains with low relief, criss-crossed by rivers and water channels and surrounded by tidal marshals and swamps. The impact of urbanization in Khulna in terms of mass poverty, gross inequality, high unemployment, under-employment, and proliferation of slum areas and squatters and general deterioration in overall environmental conditions have become the major concerns of the policy issues (Akter *et al*. 2008). There is clear evidence that water is in short supply, there is unhygienic sanitation conditions and high incidence of diseases. Because of limitation of time and resources, it was impossible to survey the whole of the city.

All these waste are dumped in the Rajbandh disposal site in order to further treatment and the residue is disposed in the sanitary landfill. For this reason, to understand the solid waste management and the tradition means of response of the people towards solid waste management at Rajbandh in KCC is selected. Location of Rajbandh is shown in the map in Figure 1.



Map-1**:** Bangladesh

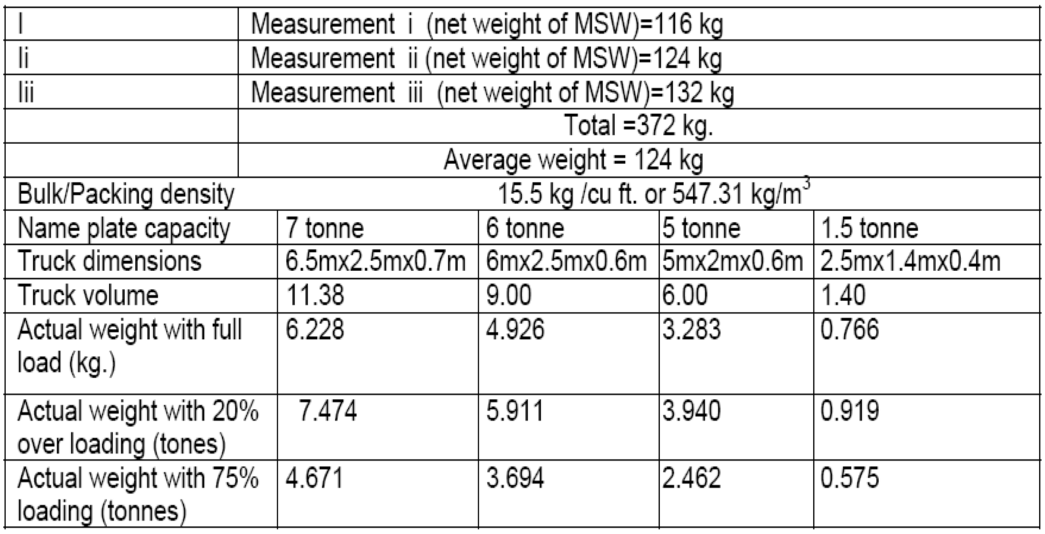
Map-2**:** Khulna City Corporation

Figure 1: Location of Rajbandh Waste Disposal Site

Rajbandh Trenching Ground Site

It is the only official dumping site at present over 25 acres in area situated at a distance of about 10 kms to the west of KCC Headquarter. Field investigators were stationed at the dumping sites to collect information on the nos. of trucks, capacity measurement of trucks, nature of solid waste and the origin of the waste. Four field investigators were engaged there from 6.00AM to 12:00 midnight for actual waste transportation. 16-19 waste carrying trucks were employed per day at Rajbandh Trenching Ground. Table 1 describes about the bulk density data of solid waste of Khulna cily.

Table 1: Bulk/ Packing density data of Khulna city solid waste and actual weights of MSW per truck



I	Measurement i (net weight of MSW)=116 kg			
Ii	Measurement ii (net weight of MSW)=124 kg			
Iii	Measurement iii (net weight of MSW)=132 kg			
	Total =372 kg.			
	Average weight = 124 kg			
Bulk/Packing density	15.5 kg /cu ft. or 547.31 kg/m^3			
Name plate capacity	7 tonne	6 tonne	5 tonne	1.5 tonne
Truck dimensions	6.5mx2.5mx0.7m	6mx2.5mx0.6m	5mx2mx0.6m	2.5mx1.4mx0.4m
Truck volume	11.38	9.00	6.00	1.40
Actual weight with full load (kg.)	6.228	4.926	3.283	0.766
Actual weight with 20% over loading (tones)	7.474	5.911	3.940	0.919
Actual weight with 75% loading (tonnes)	4.671	3.694	2.462	0.575

COLLECTION OF GROUNDWATER SAMPLES

The water sample was collected from shallow tubewells (around 90~130 ft deep) surrounds one km radius of dumping area. This water sample was taken into the laboratory for examining the water quality whether it is suitable or not. Then water samples around two and three km radius were collected and thus the relationship among one, two and three km radius data has been established.

Analytical Methods

Laboratory determination of BOD_5 was accomplished using membrane electrode DO meter (HACH, USA). For the determination of COD, closed reflux method using $K_2Cr_2O_7$ oxidizing agent was used. The determination of pH and Conductivity were done using electrodes (HACH, USA). The Hardness (as $CaCO_3$) test was performed using EDTA standard methods. For TDS in water sample was determined at 105^0C using laboratory oven. Cloride ion concentration in water sample was determined using standard titration methods. For Iron (Fe) test ferrover iron reagent was used with UV Spectro-photometer (DR 2500, HACH, USA). For determination of Nitrate (NO_3) and phosphate (PO_4) Nitrover and Phosver reagents were used (Wickramasinghe et al. 2004).

RESULTS AND DISCUSSIONS

Mismanagement Practices

Wastes are dumped near the road sites thus foul odor are emitted which pollute the environment. Blockage of drainage systems occurs for wastewater overflow during rainy season due to dumping the waste at un-official site. This causes pollution of surface water bodies as well as groundwater. Moreover, wastes are spread of by scavenging birds and animals sometimes. Indiscriminate disposal of hospital wastes that contain pathogenic organisms, may lead to spread of infectious diseases. Due to this occurrence, transmission of vector-borne diseases happens.

Figure 2: Main constraints in the existing management system

Due to poor transportation facilities rutting is formed during rainy season when heavy loaded truck is moving over the muddy road. The waste water is accumulated on the rutting and egress into the groundwater thus contamination is occurred. In the burning chamber hospital waste as well as non biodegradable waste is also burned. So as a result various chemical compounds are emitted in the environment and thus air is contaminated. Specified temperature is not maintained in this burning chamber. The total existing mismanagement practices are shown in details in figure 2.

Groundwater Quality Around Rajbandh Landfills

Biochemical Oxygen Demand (BOD) and Chemical Oxygen Demand (COD)

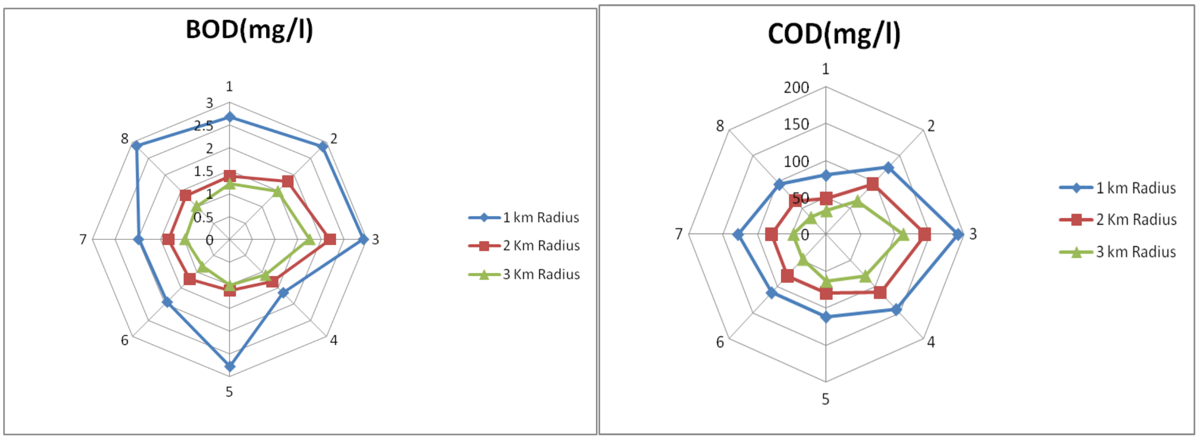


Figure 3: Graphical representation of BOD and COD around Rajbandh waste dumping site

A high concentration of organic matter was found in the collected water samples. Maximum biochemical oxygen demand (BOD) value around one, two and three km radius is 2.92, 2.19 and 1.75 mg/l respectively. In Bangladesh standard BOD_5 is 0.2 mg/l. Large amount of COD in water indicate the low quality of groundwater. In Bangladesh, standard COD_5 is 4 mg/l. The above graphical representations of BOD and COD shows that the values are less in one km but it is rising gradually in two and three km radius (Figure 3). So it is a clear indication that leachate contaminates the groundwater.

Nitrate and Phosphate

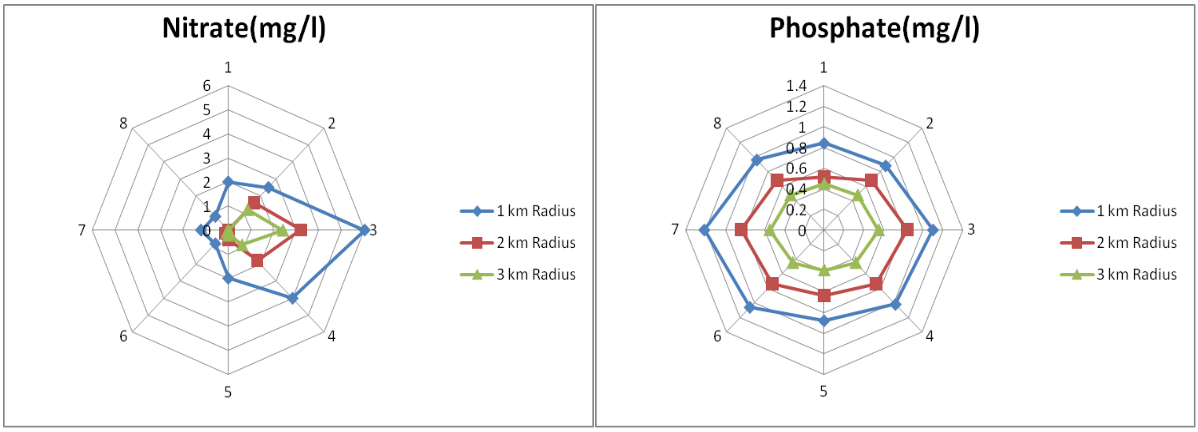


Figure 4: Graphical representation of Nitrate and Phosphate around Rajbandh waste dumping site

The above graph of Figure 4 shows that Nitrate and Phosphate value is increasing gradually one, two and three km radius. In one km radius Nitrate and Phosphate is less means the quality of groundwater is low because of having position near the waste dumping site. This value is gradually raising means groundwater is less affected by leachate because of having longer distance from waste dumping site. High concentrations of Nitrate (3.2 mg/L) and Phosphate (1.2 mg/L) were noticed in the shallow groundwater within 1 km radius of the dumping site; which suggests the possible contamination of groundwater aquifer due to leachate.

Chloride Content

For drinking purpose, Bangladesh standards Chloride value is 150-600 mg/l (Ahmed and Rahman, 2000). WHO guideline value is 250 mg/l. Figure 5 shows that the Chloride value is below the Bangladesh standard in one km but it is rising gradually in two and three km radius. High concentrations of Chloride content (800 mg/L) was noticed in the shallow groundwater within 1 km radius of the dumping site; which suggests the possible contamination of groundwater aquifer due to infringement of leachate beneath the landfills.

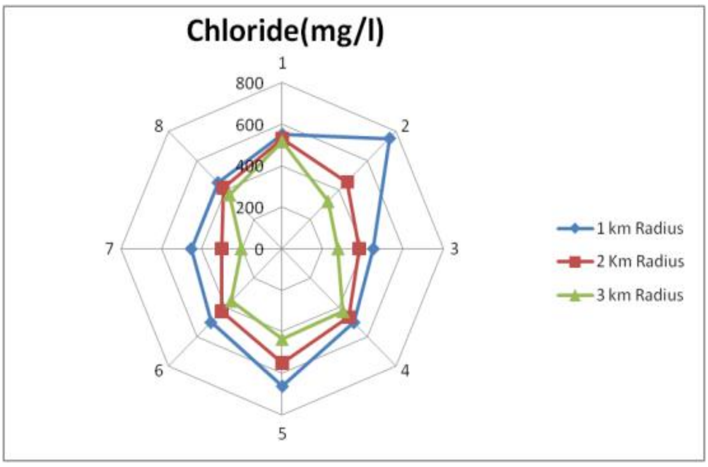


Figure 5: Graphical representation of Chloride around Rajbandh waste dumping site

Some other physicochemical parameters such as pH, Conductivity, Alkalinity, Hardness, Total Dissolved Solid (TDS), etc. were also investigated to depict a comprehensive scenario of the shallow groundwater aquifer around the disposal site. The summary of groundwater quality around Rajbandh site according to 1 km, 2 km and 3 km distances is shown in Table 2, 3 and 4.

Table 2: Shallow groundwater around one km radius of Rajbandh landfill site

Location	BOD (mg/L)	COD (mg/L)	Conductivity (µs/cm)	Alkalinity (mg/L)	Hardness (mg/L)	TDS (mg/L)	Chloride (mg/L)	Iron (mg/L)	pH	Nitrate (Mg/L)	Phosphate(Mg/L)
N	2.68	80	1286	245	115	900	550	0.75	6.84	2	0.84
NE	2.87	128	1283	240	148.2	1000	750	0.08	6.98	2.5	0.88
E	2.92	192	1086	265	175	900	450	0.01	7.08	6	1.1
SE	1.65	144	1030	255	155	800	500	0.05	7	4	1.02
S	2.78	112	1104	275	188	650	660	0.05	7.16	2	0.88
SW	1.94	112	1321	230	201.9	1000	500	0.23	6.94	0.8	1.06
W	2	128	1066	270	135	600	450	0.08	7.2	1.2	1.21
NW	2.89	96	1126	235	129.6	900	450	0.01	7.03	0.8	0.96

Table 3: Shallow groundwater around two km radius of Rajbandh landfill site

Location	BOD (mg/L)	COD (mg/L)	Conductivity (µs/cm)	Alkalinity (mg/L)	Hardness (mg/L)	TDS (mg/L)	Chloride (mg/L)	Iron (mg/L)	pH	Nitrate (Mg/L)	Phosphate(Mg/L)
N	1.39	48	1068	215	88	650	530	0.31	6.63	0	0.51
NE	1.79	96	972	232.5	116.5	735	455	0.24	6.75	1.6	0.67
E	2.19	144	876	250	145	820	380	0.17	6.88	3.2	0.84
SE	1.31	112	929	215	116.5	685	465	0.32	6.95	1.8	0.74
S	1.12	80	982	240	88	550	550	0.48	7.03	0.4	0.64
SW	1.23	80	971	225	96.5	515	425	0.78	6.84	0.2	0.74
W	1.34	80	960	210	105	480	300	1.09	6.86	0	0.84
NW	1.36	64	1014	212.5	96.5	565	415	0.7	6.74	0	0.67

Table 4: Shallow groundwater around three km radius of Rajbandh landfill site

Location	BOD (mg/L)	COD (mg/L)	Conductivity (µs/cm)	Alkalinity (mg/L)	Hardness (mg/L)	TDS (mg/L)	Chloride (mg/L)	Iron (mg/L)	pH	Nitrate (Mg/L)	Phosphate(Mg/L)
N	1.22	32	867	187	68	425	515	0.26	6.53	0	0.45
NE	1.49	64	782	205	95	475	320	0.13	6.68	1.2	0.48
E	1.75	112	596	225	127	520	276	0.04	6.74	2.4	0.55
SE	1.11	80	796	186	79	415	425	0.25	6.84	0.85	0.45
S	1.01	64	782	215	61	355	435	0.11	6.25	0.2	0.39
SW	0.85	48	698	201	78	375	355	0.13	6.71	0.1	0.45
W	0.98	48	768	182	69	298	202	0.03	6.69	0	0.55
NW	1.03	32	889	178	78	325	368	0.04	6.55	0	0.48

CONCLUSIONS AND RECOMMENDATIONS

The major findings of this study are outlined as below:

- The shallow groundwater quality was found to be gradually deteriorated in three, two and one km radius around Rajbandh landfill site. Maximum biochemical oxygen demand (BOD_5) was obtained 2.92 mg/l. In Bangladesh, drinking water standards for BOD_5 is 0.2 mg/l. The value of other water quality parameters such as Chemical Oxygen Demand (COD), Electrical Conductivity, Alkalinity, Hardness, Total Dissolved Solid (TDS), Chloride Content, Iron, pH, Nitrate and Phosphate were also found to get worse progressively towards the landfills. So, it is a clear indication that the groundwater is contaminated due to leachate.

- Misplacement of solid wastes around Rajbandh dumping site, unplanned transportation facilities, inadequate lining facility, and improper burning of wastes were the main constraints in the existing management system.
- Strict law should be enforced to ensure sustainable management practice of Rajbandh landfills site. In this regard, good transportation facilities using rigid or flexible pavement with appropriate roadway capacity instead of muddy road would be effective. An appropriate incinerator for hospital and other hazardous waste should be provided. Toxic polyvinyl chloride, polythene, etc. should be screened out prior to incineration to avoid the release of harmful chemical components. Proper lining arrangement with frequent monitoring seems imperative for the protection of groundwater from leachate contamination of solid wastes landfills.

In general, the outcome of this study would provide a datum for any future attempt concerning the solid waste management and surrounding groundwater protection for the Rajbandh landfill site.

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Crack Behavior of Composite Clay as Landfill Cover Liner

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ABSTRACT

Composite clay can be one of the best solutions as the top lining materials in the economic point of view. The aim of the work was to check the crack and shrinkage behavior of the composite clay as landfill cover liner. For each of different percentages like 5%, 10%, 12.5%, 15% and 20% of brick chips was mixed with clay and then average crack width and shrinkage was determined. It was found that among various ratios, 20% brick chips ratio showed preferable performance.

INTRODUCTION

Landfill Cover is one of the most important components for the safe design of landfill system by protecting the whole system in post closure period from external atmosphere and even during the operation period. The main intention of cover is to oppose the forces of nature and prevent water from moving downward in response to the force of gravity (Victor, 2008). After filling of landfill being completed, a final cover layer is provided which usually consists of compacted clay, composite clay, geo membrane or so on. In some cases, above the compacted clay the geo textile is placed and solid wastes are dumped beneath the compacted clay. It is used to prevent the infiltration of rainwater. The rainwater can be mixed with waste and produce leachate due to biochemical reaction. Leachate is a very much hazardous liquid. By preventing infiltration of rainwater leachate production can be reduced on a great extent. Top liners also prevent the migration of landfill gases.

Geo membrane or geo textile shows better efficiency as top liner material. It is expensive to use as top lining material. Brick chips, stone chips or other materials can be mixed with naturally available clay. Brick and stone chips are the materials of non shrinkage and posses sharp enough surface area for suitable adhesion. Besides, they have the sharp angles.

Soils classified as inorganic clay with high plasticity (CH) is considered as the suitable material for landfill liner (Oweis & Khera 1998). If naturally available clay or clayey soil is not suitable for liner, kaolinite or commercially available high swelling clay such as Bentonite can be mixed with local soils or sand. In Bangladesh these materials are not locally available and would have to be imported from elsewhere and could significantly increase the cost of construction (Alamgir et al. 2005). Clay is very efficient to mix with aggregate to create a perfect bonding. The shrinkage property of clay can be reduced by aggregate in considerable extent. It is called composite clay. Composite clay is economical rather than geo membrane or geo textile. Formation of crack is the main problem while composite clay is used. The study is based on the investigation of crack behavior of composite clay with respect to time and drying. Shrinkage is also a fatal problem in using composite clay as top liner. So the objectives of this study can be pointed out as-

- To check crack behavior of composite clay as landfill cover liner.
- To determine the optimum content of brick chips for composite clay as landfill cover liner to have lesser cracks.

MATERIALS AND METHODS

This work is based on the preparation and casting of mold and monitoring of propagation of crack with operation. The crack is distributed along the longitudinal direction due to shrinkage. The emission of soil sample increases with time. It is also responsible in crack formation. The soil sample in mold was subjected to several number of cracks. These distributed cracks are recorded within a time interval (2 or 4 days). The soil samples in molds are placed under dry condition so that the evaporation can readily take place. The formation of crack is time dependant. The emission of moisture from soil sample causes shrinkage of soil.

The work was done in a sequential manner and included several stepwise works with time. The total work was done was in such a manner that it can be adjusted with practical application.

- Firstly, the soil sample was collected from a location nearby the students' residential hall Khan Jahan Ali hall. This site was easily accessible and the sample could be collected in any season except the rainy season.
- The initial moisture content in the sample was measured.
- Various soil tests as standard Proctor test for optimum moisture content, Atterberg limits, Specific gravity, Grain size analysis, Grain size analysis by Hydrometer method were accomplished. Also Sieve analysis of brick chips was conducted.
- Four (04) wooden planks were assembled to form a mold.
- These molds were rectangular in shape.
- The soil sample was placed with brick chips in such a condition so that it achieved a form like a liner.
- For 25% moisture content three (03) samples in the mold of dimension 30 cm*8 cm*6 cm were prepared.
- The brick chips were mixed with clay at various percentages. The brick percentages with clay were 5%, 10%, 12.5%, 15%, 20% and 20%.
- For each percentage, three (03) samples were prepared.
- Then those samples were brought outside so that they got dried.
- The formation of crack was monitored by recording data of crack width.
- The data of shrinkage on all four (04) sides of samples were recorded.
- Then various percentages of brick chips were compared with control samples.
- Finally, the percentage of brick chips having comparatively lesser crack was determined from further analysis.

RESULTS AND DISCUSSION

All the results are shown in the following tables by comparing the data of different percentages of brick chips with clay. The data of shrinkage of soil sample with brick chips in molds are also shown in tables. The variation of different data is also presented in graphs.

Table 1 Variation in crack width and shrinkage for 0% brick chips

Time (days)	Variation in Crack width (cm)			Variation in Shrinkage (cm)		
0	0.00	0.00	0.00	2.00	2.20	2.05
1	0.20	0.15	0.22	2.15	2.40	2.15
6	0.30	0.25	0.26	2.20	2.40	2.25
9	0.36	0.30	0.30	2.30	2.50	2.30
12	0.36	0.35	0.30	2.75	2.85	2.80
15	0.40	0.40	0.40	2.85	2.95	2.90
19	0.45	0.50	0.45	2.95	3.00	3.00
25	0.45	0.50	0.45	3.20	3.50	3.00

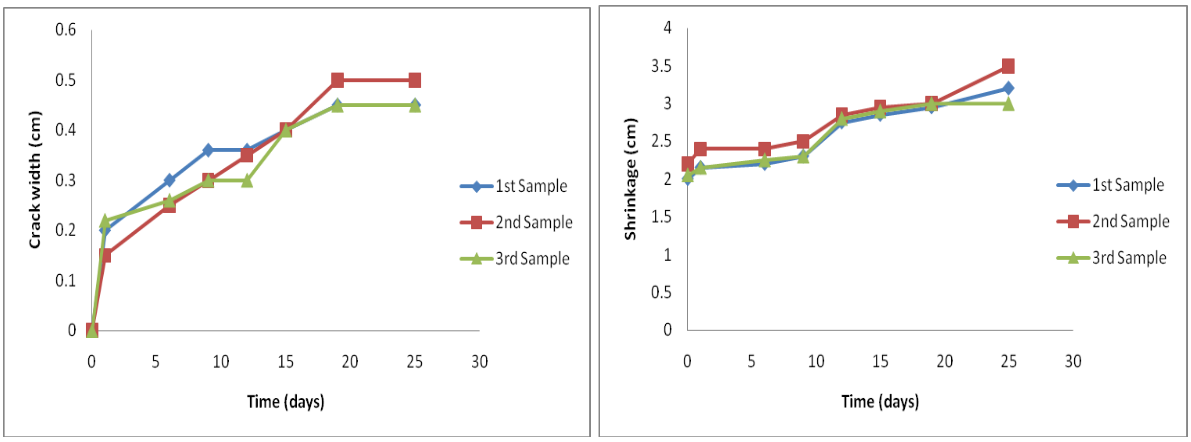

Figure 1 Variation of crack width and shrinkage with time for 0% brick chips

Table 2 Variation in Crack width for 5%, 10% and 12.5% brick chips

Time (days)	Variation in Crack width for various percentages of brick chips (cm)								
	5%			10%			12.5%		
0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1	0.20	0.60	0.40	0.60	0.60	0.50	0.00	0.00	0.00
6	0.20	0.60	0.40	0.90	0.75	0.50	0.00	0.00	0.00
9	0.25	0.60	0.40	0.90	0.75	0.50	0.20	0.24	0.20
12	0.30	0.60	0.40	1.00	1.00	0.50	0.25	0.30	0.25
15	0.30	0.60	0.50	1.00	1.00	0.50	0.30	0.40	0.30
19	0.30	0.60	0.60	1.00	1.00	0.50	0.35	0.40	0.35
25	0.30	0.60	0.60	1.00	1.00	0.50	0.35	0.40	0.40

Table 3 Variation in Shrinkage for 5%, 10% and 12.5% brick chips

Time (days)	Variation in Shrinkage for various percentages of brick chips (cm)								
	5%			10%			12.5%		
0	0.95	1.00	0.80	1.00	0.80	0.85	0.18	0.15	0.17
1	2.80	2.80	2.40	2.60	2.60	2.60	0.40	0.40	0.35
6	2.85	2.80	2.55	3.00	2.60	2.80	0.50	0.55	0.55
9	2.85	2.85	2.55	3.00	2.60	2.95	0.70	0.80	0.80
12	2.90	2.90	2.65	3.00	2.50	2.90	0.95	1.00	1.05
15	2.90	2.90	2.70	2.90	2.50	3.00	2.20	2.00	2.00
19	2.90	2.90	2.65	2.80	2.60	3.25	2.25	2.30	2.30
25	2.90	2.90	2.65	2.80	2.60	3.25	2.55	2.60	2.60

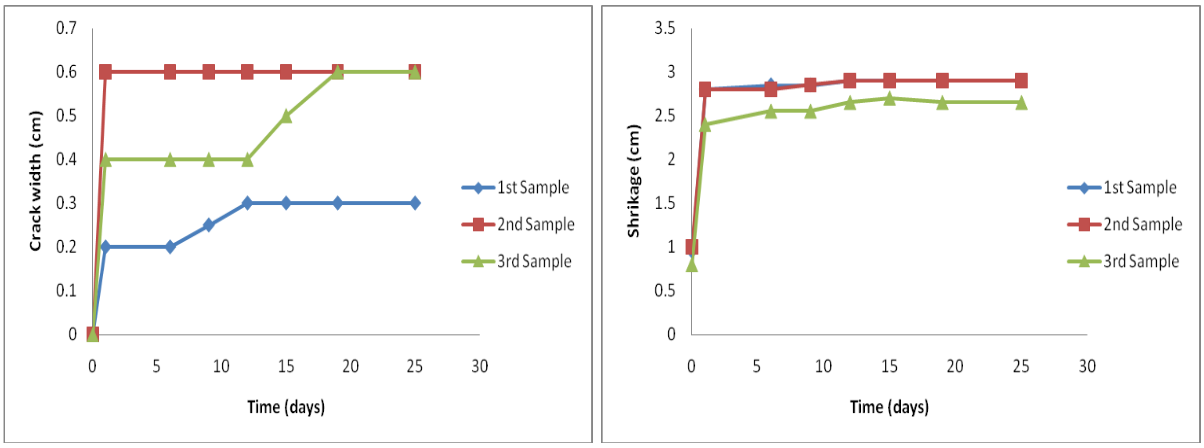

Figure 2 Variation of crack width and shrinkage with time for 5% brick chips

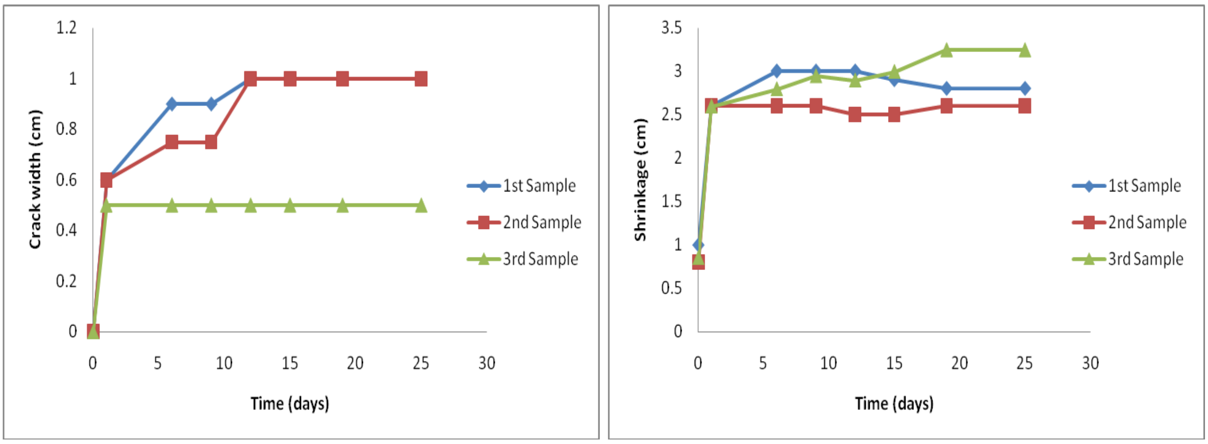

Figure 3 Variation of crack width and shrinkage with time for 10% brick chips

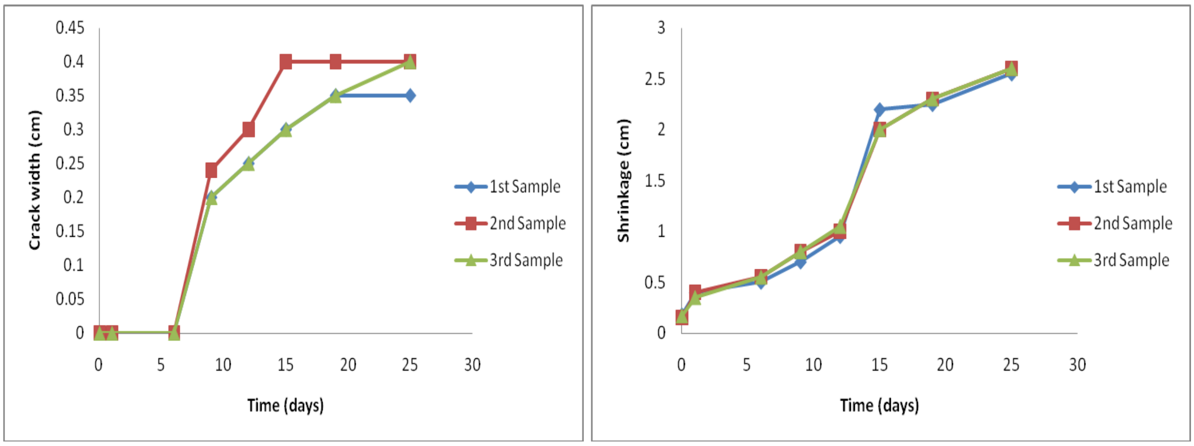

Figure 4 Variation of crack width and shrinkage with time for 12.5% brick chips

It is clear from the above tables that at 12.5% brick chips, the amount of crack width was less than the crack in the control mold sample. Also after 25 days, the total shrinkage in the mold with clay with brick chips 5%, 10%, 12.5% were 2.0 cm, 3.25 cm and 2.6 cm. These shrinkage values were below the shrinkage found in the control mold samples. The variations of crack width and shrinkage with different brick chips percentages are shown in Figure 2, 3 and 4. In almost every percentage, the crack width and shrinkage values increased with time from initial small values.

Table 4 Variation in Crack width for 15%, 20% and 25% brick chips

Time (days)	Variation in Crack width for various percentages of brick chips (cm)								
	15%			20%			25%		
0	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1	0.50	0.50	0.85	0.20	0.30	0.25	0.00	0.00	0.00
6	1.00	0.70	0.85	0.20	0.30	0.30	0.00	0.00	0.00
9	1.00	0.70	1.20	0.20	0.35	0.30	0.15	0.23	0.20
12	1.10	0.70	1.30	0.20	0.40	0.40	0.20	0.28	0.20
15	1.10	0.70	1.30	0.20	0.40	0.40	0.30	0.35	0.30
19	1.10	0.80	1.30	0.20	0.40	0.40	0.40	0.45	0.50
25	1.10	0.80	1.30	0.20	0.40	0.40	0.40	0.50	0.60

Here less amount of crack 0.4 cm was found at 20% brick chips with respect to control mold sample.

Table 5 Variation in Shrinkage for 15%, 20% and 25% brick chips

Time (days)	Variation in Shrinkage for various percentages of brick chips (cm)								
	15%			20%			25%		
0	0.70	0.60	0.70	0.00	0.00	0.00	1.40	1.40	1.30
1	2.25	2.40	1.80	2.05	2.00	2.00	1.50	1.40	1.50
6	2.30	2.60	2.10	2.05	2.00	2.00	1.65	1.50	1.60
9	2.20	2.60	2.15	2.05	2.10	2.00	1.70	1.60	1.70
12	1.95	2.60	2.10	2.20	2.05	2.00	1.95	1.85	1.80
15	2.00	2.70	2.10	2.00	2.05	2.00	2.05	1.95	1.95
19	2.00	2.75	2.00	2.00	2.00	2.00	2.20	2.05	2.05
25	2.00	2.75	2.00	2.00	2.00	2.00	3.20	3.00	3.15

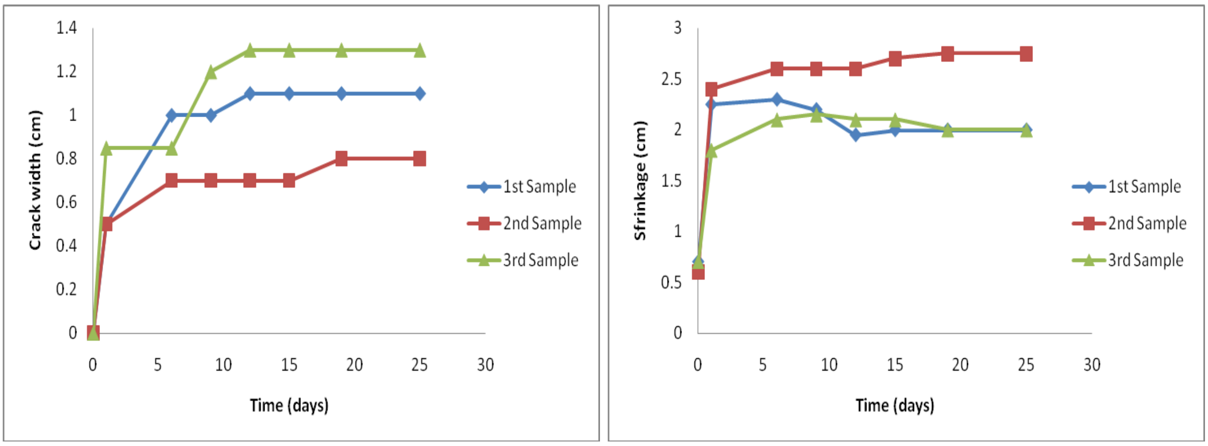

Figure 5 Variation of crack width and shrinkage with time for 15% brick chips

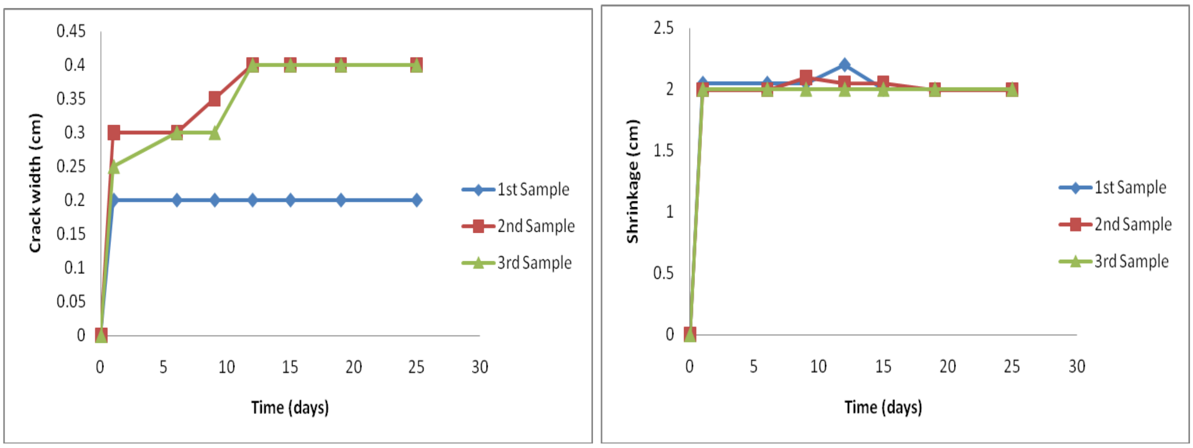

Figure 6 Variation of crack width and shrinkage with time for 20% brick chips

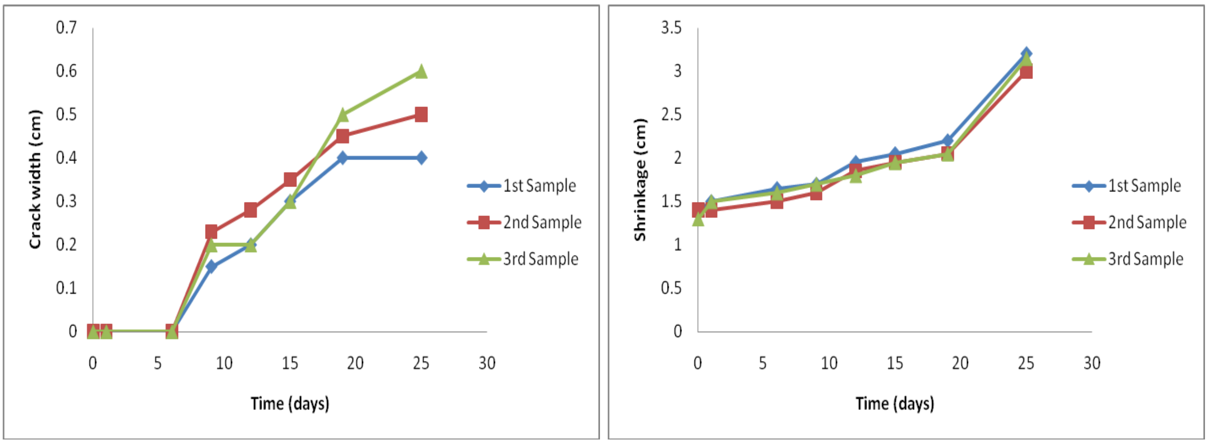

Figure 7 Variation of crack width and shrinkage with time for 25% brick chips

After 25 days, the amount of shrinkage for 15%, 20% and 25% brick chips was 2.75 cm, 2.00 cm and 3.20 cm, respectively. Those values were below the shrinkage found in control sample. 20% brick chips showed considerably less amount of shrinkage.

Average amount of crack width and shrinkage values for different percentages of clay after 25 days can be represented in a single table as-

Table 6 Variation in average Crack width and Shrinkage

Percentage of Clay (%)	Average Crack width (cm)	Average Shrinkage (cm)
0 (Control sample)	0.47	3.23
5	0.50	2.81
10	0.83	2.88
12.5	0.38	2.58
15	1.06	2.25
20	0.33	2.00
25	0.57	3.12

CONCLUSION

The mold with only clay samples showed considerable variation in crack width and shrinkage with respect to time and drying. After 25 days, average crack width and shrinkage was determined as 0.47 cm and 3.23 cm, respectively. With the increase of clay percentage, crack width and shrinkage values showed variation. Both the amount of crack and shrinkage were less than control sample at 12.5% and 20% clay samples. The most preferable percentage of clay according to amounts of crack and shrinkage was found as 20%.

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Influence of Cap Liner on Leachate Characteristics: Results from Pilot Scale Landfill Lysimeter

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ABSTRACT

Investigating the leachate characteristics influenced by different soil covers and hence to determine the best applicable soil in sanitary landfill as top cover is the principal objective of this study. To this endeavor, four lysimeters at varing operational conditions were simulated. Among four, one was open dump i.e. filled solely with municipal solid waste (MSW) having no cover soil and rests were sanitary landfill lysimeters having three different types of soil covers i.e. sandy loam soil, silty loam soil and clay soil. Study was conducted in the rainy sesson. Results depicted that open dumped lysimeter produced more quantity of leachate than sanitary landfills. Waste settlement rate was the maximum in lysimeter having no top cover. Pollutants quantities were found maximum in the open dump lysimeter, compared to the sanitary landfill lysimeters. Lysimeter having sandy loam soil was proved to be beneficial in reduction the amount of all pollutants from leachate.

INTRODUCTION

Landfill is a unit operation for final disposal of ‘municipal solid waste’ (MSW) on land. Open dumping and sanitary landfill are two types of landfill. The open dump approach still remains the predominant waste disposal alternative in developing countries erectionting noteworthy nuisance and environmental problems. With the accelerated generation of waste caused by increasing population, urbanization, and industrialization, the problem has become even worse. In South and South East Asia more than 90% of Municipal Solid Waste (MSW) is disposed of in open dumps (Rafizul et al. 2009b). Sanitary Landfills are the most popular method of ultimate disposal of solid waste. It is a land disposal site employing an engineered method of disposing of solid waste on land in a manner that minimize volume, and applying and compaction cover material at the end of each day (Davis and Cornwell, 1998). Lysimeter is a simulated from of sanitary landfill in the sense of control device. The word lysimeter is a combination of two greek words “Lusis” means “Solution” and “Metron” means “Measure” and orginal aim is to soil leaching (Rafizul et al. 2009a). Leacahte is the most polluted liquid generated in a landfill due to the water content that enters the landfill from external sources, surface drainage, rainfall, groundwater, and water from waste material. The leachate generated from MSW disposal sites is considered as one of the highly contaminated resources from physical, chemical and biological point of view. However, the best possible knowledge of leachate characteristics at a specific site is an essential management tool (Rafizul et al. 2012). This is not only important for new contaminated needs designed in advance, where leachate will be extracted, but also important for the old landfill where the environmental safeguards rely on the natural attenuating properties of the geological strata, to reduce the level of contaminant to environment (Robinson. 1989). Sanitary landfill leachate is the most complicated and costly wastewater to treat due to its high content of organic and inorganic pollutants (Bilgili et al. 2007). Proper treatment of leachate is impossible in developing countries like Bangladesh due to cost effect. Therefore it is necessary to develop strategies to reduce pollutants concentration from leachate. The cover soil used in the landfill operation might be used as one of attenuation processes to decrease the pollutants’ concentrations leached out in the leachate (Karnchanawong et al. 1995). Hence, the study will allow having an outcome for low cost solid waste disposal technology and that will be sustainable also, on the basis of effects of soil cover type on characteristics of leachates generated from landfill lysimeters.

MATERIALS AND METHODS

The detailed procedure for the design of reference cell, set up of four pilot scale landfill lysimeter at KUET campus, characterization of MSW and soil deposited in each lysimeter as well as the analytical methods of leachate were presented and hence discussed in the following articles.

Set-up of Landfill Lysimeter

Four lysimeters made of PVC pipe were prepared in this study. Among four, one lysimeter simulated as open dump i.e. filled solely with MSW having no top cover soil designed as lysimeter-L. In this lysimeter the MSW was not covered by a top cover system to pervert the movement of air, water and generated landfill gas (LFG). Moreover, the thickness of the deposited MSW in lysimeter-L is such that it is expected the atmospheric air can move in the entire MSW deposited in this cell with negligible inference. Due to the mentioned practical situations, lysimeter-L, represents the aerobic condition.

In contrary, the other three lysimeters were treated as sanitary landfill having three different types of soil covers i.e. sandy loam soil, silty loam soil and clay soil and hence designated lysimeter-LSDL, lysimeter-LSTL and lysimeter-LC, respectively. These three landfill lysimeter operated as an anaerobic conditions in presence of cover soil. The cross section of reference cell with in detailed for each lysimeter shown in Figure 1. The height and inner diameter of all lysimeters were 1800 mm and 200 mm, respectively. The upper 100 mm free space is for adding rainfall. The lower part containing a gravel layer served as the waste base and allowed the leachate to flow through the collecting pipe. A geo-textile sheet was used to avoid rapid clogging of the under laying pipe. The operational conditions of all the concerned lysimters were presented in Table 1. In addition, using the textural classification chart (Figure 2) the soils used as top cover were classified shown in Table 2. Moreover, the mass density of soil used as top cover in each lysimeter was found as 1590 kg/m^3.

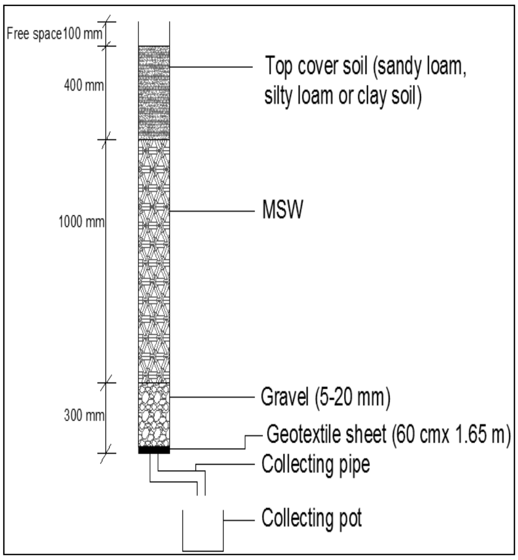

Figure 1 Cross section of reference cell

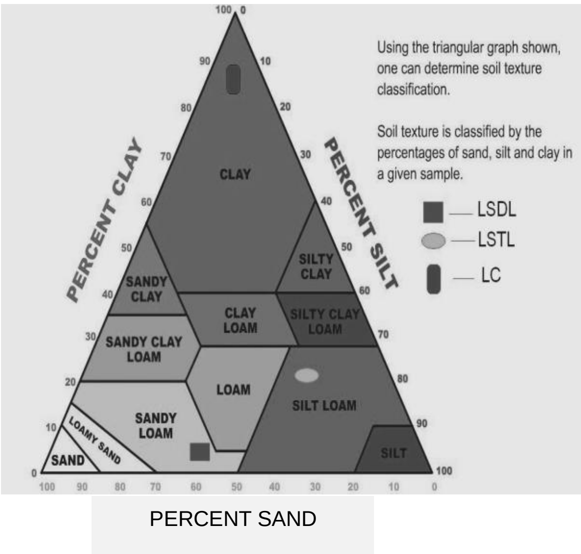

Figure 2 Classify soil used as top cover using textural classification chart (U.S. Department of Agriculture)

Table 1 Operational conditions used in lysimeter to simulate different landfill conditions

Designed lysimeter	Operational condition	Refuse (Kg)	Cover soil specification
Lysimeter-LC	Applicability of designed cover soil	22	clay
Lysimeter-LSDL			Sandy loam
Lysimeter-LSTL			Silty Loam
Lysimeter-L	Present practice of open dump		Without top cover

Table 2 Classification of soil used as top cover in this study

Parameters	Lysimeter-LSDL	Lysimeter-LSTL	Lysimeter-LC	Analy method
Moisture content (%)	15	30.9	39.7	ASTM D-2216-90
pH	8.18	8.31	7.63	-
Specific gravity	2.67	2.55	2.51	ASTM D-854
Sand: Silt: Clay	59:36:5	24:55:21	5:7:88	ASTM C-136, D-422
Soil classification	Sandy Loam	Silty Loam	Clay	USDA

Characterization of MSW Deposited in Landfill Lysimeter

The quantity and degree of contamination of leachate depends on physical and chemical characteristics of MSW, rate of degradation of organic fraction from MSW, degree of compaction, height of waste and therefore age of MSW in deposited landfill (Alamgir et al. 2006). To this attempt, before filling the MSW, the MSW was characterized according to their physical composition and hence described as follows.

3 Kg of MSW was taken and hence sorting all the composition consists of 82.2 (w/w) of food/vegetable, 8.3 (w/w) of paper and paper products, 2.6 (w/w) of plastic and polythene, 1.3 (w/w) of textile and wood, 1 (w/w) of rubber and leather, 3 (w/w) of dust, ash and mud products and 1.6 (w/w) of others (rope, brick, glass, tin etc.). Here, the predominant component is food and vegetables waste having high organic content on which quality of leachate primarily depends on. Moreover, the organic content, moisture content, pH, volatile solids (VS) and ash residue of MSW were 51 %, 66 %, 7.62, 57 % and 45%, respectively. The moisture content of MSW was measured in the laboratory by incinerated in the electric oven at 105^{o}C for 24hr. Moreover, for determining the organic content MSW was incinerated in the muffle furnace at 550^{o}C for 5 hr. The detailed procedure for measuring the moisture and organic content of MSW in lysimeter can be obtained in Austrian Standard (S 2023). pH was determined by pH meter (HACH, Model No. Sens ion 156), volatile solid by ignited MSW at 950^{0}C, after burning the MSW ash residue was found. Moreover, the waste density filled in each lysimeter was 700 kg/m^{3}. All the findings of MSW are agreed with a feasibility study conducted on characteristics of MSW by WasteSafe (2005) project in Khulna city, Bangladesh.

Rainfall Addition, Leachate Sampling and Analysis

Bangladesh is a country of six seasons. But in this study only the rainy season was considered when the leachate generation rate is the maximum. The duration of rainy season in Bangladesh is from mid June to mid September. The daily precipitation data from June 2010 to August 2010 of Khulna city were used to replicate rainfall in this study. The study was conducted for 5-month period. For the first 3-months, June to August daily precipitation data was used and for the next 2-months, July to August precipitation data was used. To work out actual infiltration through the landfill, a runoff coefficient of 0.22 (for compacted top soil with a slope of 3%) and an evaporation level of 28 % of the total rainfall estimated by Jica et al. (1992) were used. The infiltration through the landfill was estimated to be 50 % of daily rainfall. Distilled water with an amount equal to 50 % of the daily-recorded rainfall in 2010 was fed once a week into each lysimeter by adding all the daily rainfall of that week. Leachate quantity, waste settlement rate and leachate characteristics i.e. pH, electric conductivity (EC), total solids (TS), sulfate (So_4^{2-}), chloride (Cl^-), alkalinity as $CaCo_3$, COD, BOD_5 and Total Kjeldahl Nitrogen **(**TKN) were determined once a week.

Analytical Methods for the Appraisal of Leachate

In the laboratory, pH was determined by pH meter (HACH, Model No. Sens ion 156), EC by conductivity meter (HACH, Model No. Sens ion 5), chloride by potentiometric titration method using silver nitrate solution, alkalinity by titration method, COD by closed reflexive method as per the standard methods (APHA, 1998) as well as BOD_5 by dilution method (titration). In addition, TS dried at 103-105 ^{0}C, sulfate by Sulfa Ver 4 method and TKN by macro-kjeldahl method.

RESULTS AND DISCUSSIONS

Leachate Generation

Leachate is formed in solid waste landfill when the refuse moisture content exceeds its field capacity (rainfall, initial moisture content, etc.) (Alamgir et al. 2006). Factors affecting the leachate generation from landfill are solid waste composition, initial moisture content, rainfall, evaporation and infiltration rate of rainfall and therefore the climatic condition under which the landfill is situated (Rafizul et al. 2012). Amount of rainfall added and leachate generated from different concerned lysimeters are summarized in Figure 3 over time. Cumulative amount of rainfall added into each lysimeter was 16.64 L and cumulative amount of leachate generated from lysimeters LSDL, LSTL, LC and L were 13.13, 13.66, 13.36 and 17.97 L, respectively. During the first two weeks, no rainfall was added and hence no noticeable amount of leachate was generated. At 14th day rainfall was feed into each lysimeter for the first time. For the next few days, the leachate amounts generated in a low amount even though a high amount of rainfall was added because of high void ratio in dry soil (Crites et al. 2000). Lysimeter filled solely with MSW produced the highest amount of leachate in contrast to lysimeters having cover soils although the same amount of rainfall was added to each lysimeter, due to the top cover that can reduce the percolating of rainwater (Rafizul et al. 2012). Consequently, the lysimeter having top cover can lessen the leachate quantities around 24 %, compared with the lysimeter using no cover soil.

Cumulative amounts of leachate from lysimeters having cover soils showed no any significant difference among them. The cracking of clay soil besides the blockage of particles in sandy loam and silty loam soil might result in the non significant differences of leachates' amounts generated from LSDL, LSTL and LC (Karnchanawong et al. 2009). A study conducted by Karnchanawong et al. (2009) suggested that lysimeter having no cover generated the highest amount of leachate and lysimeter having sandy loam soil as cover soil generated the lowest. Again Rafizul et al. (2012) shows that open dump lysimeter produced the utmost amount of leachate in contrast to sanitary landfill lysimeter. So the findings are convincing according to Karnchanawong et al. (2009) and also in Rafizul et al. (2012). It can be decided that, sanitary lanfill operational mode with a sandy loam soil as a top cover is proved to be most efficient in reduction of leachate quantity.

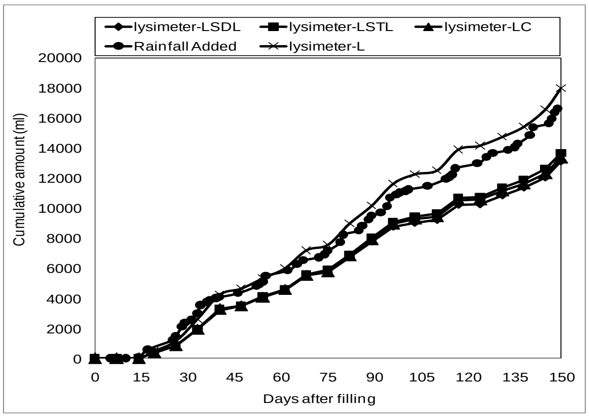

Figure 3 Cumulative amounts of rainfall added and leachate generation from lysimeter.

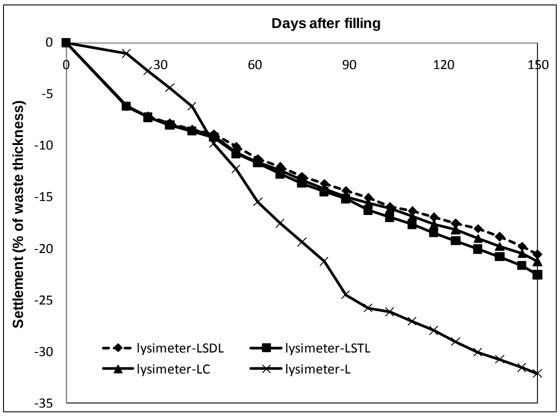

Figure 4 Variation of Settlement of MSW in lysimeter at varying operational condition.

Settlement of MSW in Landfill Lysimeter

Waste settlement in MSW landfill is the most significant problem, among all the difficulties of utilizing landfill sites for future development. The settlement mechanism in landfill is complex enough. The variation of waste composition and biological activities causes the landfill settlement in a non uniform pattern. The settlement rate of MSW deposited in lysimeters during the experimental period is summarized in Figure 4. Excessive settlement occurred in the lysimeter having no top cover in contrast to lysimeters having cover soils. The ultimate settlement amount in different lysimeters was 20.6, 22.6, 21.3 and 32.2 % (% of waste thickness) for lysimeter LSDL, LSTL, LC and L, respectively. The highest amount of settlement occurred in lysimeter "L" as the highest amount of leachate was produced from it. Here it is fascinating to note that though the highest amount of settlement occurred in lysimeter "L" the initial settlement rate were found higher for lysimeters having cover soils. As the top cover about 20 kg soil was used at the top of the waste in those lysimeters. Due to the additional weight of the soil on the waste, the waste height was compacted at the initial stage and initially settlement rate was found higher for the lysimeters having top cover. But at the latter stage waste settlement rate was increased in lysimeter "L". A study conducted by Trankler et al. (2005) and Visvanathan et al. (2002) reported that a cell with highest compaction density had the lowest settlement. El-Fadel et al. (1998) stated that rate of settlement of MSW in landfill depends primarily on the compaction of refuse, moisture content response for biodegdrdation of MSW in landfill, percolation of rainwater in refuse and the operational practices of landfill. As the compaction of waste was occurred in the sanitary landfill lysimeters initially, and no compaction was occurred in open dump lysimeter so at the latter stage, waste in open dump lysimeter settled more and produced highest amount of leachate. In accordance with Trankler et al. (2005), Visvanathan et al. (2002) and El-Fadel et al. (1998) the findings are valid.

Leachate Characteristics

The concentrations and load of leachate pollutants, generated from MSW deposited in landfill lysimeter at varying operational condition presents in Tables 3 and 4 and hence discussed in followings.

Table 3 Concentrations of leachate generated from MSW in landfill lysimeter

Parameter	Lysimeter-LSDL	Lysimeter-LSTL	Lysimeter-LC	Lysimeter-L
	Min –Max (Mean)	Min–max (Mean)	Min – Max (Mean)	Min – Max (Mean)
pH	6.41–7.86 (6.82)	6.43-7.72 (6.8)	6.37-7.67 (6.78)	6.56–7.67 (7.06)
EC	13.2–30.2 (20.46)	14.3–33.2 (21.19)	15.1–34.2 (22.97)	15.3–35.2 (23.13)
So$_4^{2-}$	590–1820 (1007)	620-1855 (1068)	620–1890 (1096)	680–2060 (1216)
Cl$^-$	700 – 3830 (2104)	610–3745 (2080)	565–3770 (2065)	510–3525 (1782)
TS	1527–11500(4272)	1592-11725(4408)	1675–11680 (4502)	1243–10960(3853)
Alkaliniy	358–8045 (3581)	305-7870 (3660)	392–8935 (3965.7)	700-10070(4464.6)
COD	2065-32305 (11368)	2580-34460 (12268)	2350–36505 (13067)	2800–39725 (14421)
BOD	1650–18220(7328)	2025-19995(7983)	2235–21060 (8677)	2650–23465(9933)
TKN	1057.9-350.5 (576)	975.3-310 (532.5)	978.7-250 (501)	702.3-145.7 (356)

Note: EC=Electric Conductivity; So$_4^{2-}$=Sulfate; Cl$^-$=Chloride; TS=Total Solid; TKN= Total Kjeldahl Nitrogen. pH has no unit; conductivity is in mS/cm; rests are in mg/L.

Table 3 Pollutant load leached from lysimeters during the study period

Pollutant load leached from lysimeter (mg/kg dry initial waste)	Lysimeter-LSDL	Lysimeter-LSTL	Lysimeter-LC	Lysimeter-L
So$_4^{2-}$	613.3	676.5	682	1000.7
Cl$^-$	1296.75	1339.4	1320.12	1489.6
TS	2706.8	2888.7	2898.8	3293.4
Alkalinity	2238.56	2275.83	2571.4	3813
COD	6938	7586	8091	11716
BOD	4410	4907	5329	8011
TKN	352.7	337.8	314	298.96

Note: So$_4^{2-}$=Sulfate; Cl$^-$=Chloride; TS=Total Solid; TKN= Total Kjeldahl Nitrogen.

pH

pH is considered as the most significant parameter that affects most of the pollutants concentration in leachate (Bilgili et al. 2007). Table 3 exhibits that average value of pH of lysimeter-L is the highest and is in alkaline range and pH of the sanitary lysimeter-LSDL, LSTL and LC is in acidic range. A study by Bilgili et al. (2007) found that pH values were higher in open dump landfill reactor compared to sanitary landfill reactor. Figure 5 illustrates that for all lysimeters pH was sharply increasing in nature for first 47 days from 6.73-7.86, 6.52-7.52, 6.68-7.93 and 6.56-7.67 and then sharply decreased up to 81 days from 7.86-6.83, 7.52-6.8, 7.93-6.75 and 7.67-7.05 for lysimeter-LSDL, LSTL, LC and L, respectively. At last stage pH was leisurely declining in nature and got a more or less unwavering state for all concerned lysimeters.

Conductivity

Figure 6 confirms that for all lysimeters, EC decreased sharply from 30 - 13.2, 32.1 - 14.3, 33.9 - 15.1 and 34.6 - 15.3 mS/cm for lysimeter-LSDL, LSTL, LC and L, respectively, upto 47 days. After that EC values were rising in condition for all concernd lysimeters. So a linear relationship between pH and EC is EC decreases with the increment of pH. With the increase of pH metal dissolution may be decreased and hence EC also decreased due to reduction of free ions. From the Table 3, mean value of EC is the utmost for lysimeter-L and lowest for lysimeter-LSDL. A similar study conducted by Karnchanawong et al. (2009) stated the same results. Hence leachate conductivity is lower through sandy loam soil in sanitary landfill operational mode as top cover.

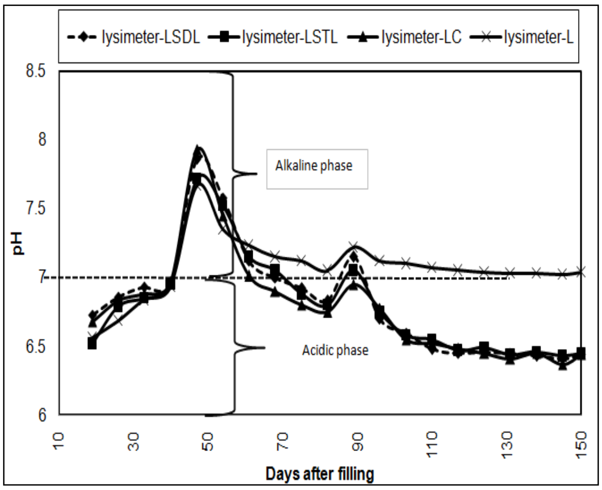

Figure 5 Variation of pH in landfill lysimeter at varying operational condition.

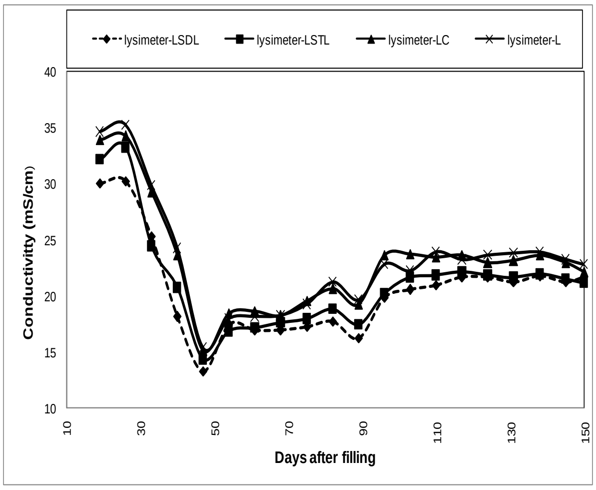

Figure 6 Variation of conductivity in landfill lysimeter at varying operational condition.

Sulfate Concentration

The mean sulfate concentration was the maximum in lysimeter-L and minimum in lysimeter-LSDL (Table 3). The Figure 7 illustrates that sulfate concentration for all lysimeters was escalating from 1540 - 1820, 1620 - 1855, 1650 - 1890 and 1810 - 2015 mg/L for lysimeter LSDL, LSTL, LC and L, respectively, up to 47 days leisurely. Concentration of sulfate primarily increased due to brisk Sulfate degradation. Sulfate degradation has a higher energy benefit for organism (Rafizul et al. 2012) which are responsible for the waste decomposition. So degradation of Sulfate initially, provided additional energy for organism that increased the waste decomposition and produced more amount of leachate at the latter stage. As a result, after 48 days unsurprisingly the concentration of sulfate in case of all lysimeters decreased due to dilution effect because of high amount of leachate generation. Moreover, the rapid decrease of sulfate is a result of predominately anaerobic condition in solid waste landfill under which sulfate is reduced to sulfide (Chian and DeWalle 1976). The uppermost cumulative amount of sulfate load during 150 days was leachated out from the lysimeter-L and lysimeter-LSDL leachated the least (Table 3). Soil can absorb different sulfate substance and sulfate ions. As a result, lysimeter using no top cover produce maximum amount of sulfate. Based on above explanation it is proved that landfill operational mode should be sanitary with a sandy loam soil as top cover.

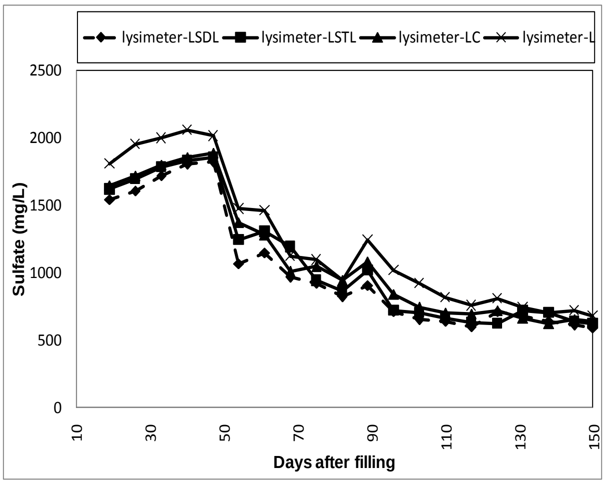

Figure 7 Variation of sulfate in landfill lysimeter at varying operational condition.

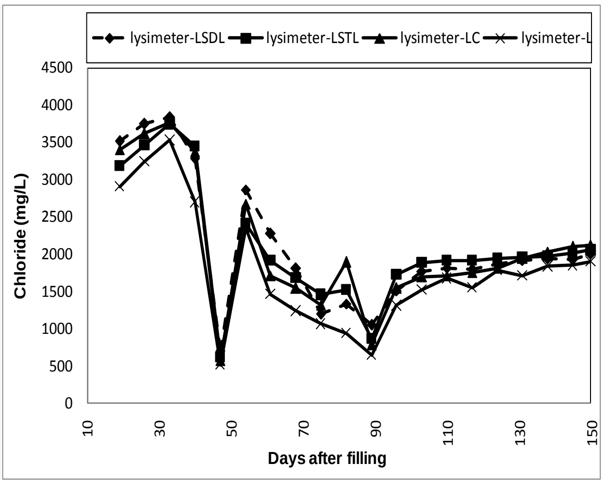

Figure 8 Variation of chloride in landfill lysimeter at varying operational condition.

Chloride Concentration

Chloride concentration was found as highest for lysimeter-LSDL and lowest for lysimeter-L (Table 3). The cover soils in different lysimeters may contain different salts and thus choride concentration is found higher in lysimeter using top cover compared to open dump lysimeter. Though the concentration of chloride is the highest for lysimeter-LSDL but cumulative amount of chloride load leachated during 150 days was the lowest for lysimeter-LSDL and highest for lysimeter-L (Table 3). The reason is that lysimeter-L produced the maximum amount of leachate while, lysimeter-LSDL produced the least. Figure 8 indicates that for all lysimeters, chloride concentration was climbing up to earliest 32 days from 3510 – 3830, 3180 – 3745, 3400 – 3770, 2910 – 3525 mg/L with the increment of pH in leachate. Manning and Robinson (1999) suggested that as a result of the increase of pH in leachate, the dissolution of chloride increases and thus the chloride in leachate also increases. So the findings are supported by Manning and Robinson (1999). After that concentration was falling up to 89 days from 3830-1045, 3745-860, 3770-780 and 3525-640 mg/L when pH values were also decreasing. Due to high quantity of leachate generation in this stage, chloride concentration may be diluted (Rafizul et al. 2012). As lysimeter operational condition with a sandy loam soil as a cover soil produced lowest amount of chloride load hence sandy loam soil is the best option to use as top cover in the landfill.

Total Solid

The average value of total solid (TS) in the leachate generated from lysimeter-L was significantly less than lysimeters-LSDL, LSTL and LC (Table 3). The higher concentrations of TS found from the lysimeters having cover soils might be because of the leaching out of the solids from the soil themselves. The Figure 9 shows that for all lysimeters TS values were found increasing for the first three sampling and then decreased harshly up to 90 days from 11500 - 1527, 11725 -1674, 11680 - 1778 and 10960 - 1243 mg/L for lysimeters-LSDL, LSTL, LC and L, respectively. TS concentration was decreased due to dilution effects and reduction in availability of loose materials with the increase of landfill. Afterward the graph was gotten a stable state. As TS load was found highest in lysimeter-L and lowest in lysimeter-LSDL (Table-4). So sandy loam soil is the best soil as top cover. A study conducted by Karnchanawong et al. (2009) stated the same result.

Alkalinity

The alkalinity of water is due primarily to salts of weak acids and strong bases (Rafizul et al. 2009b). Mean concentration as well as alkalinity load is the maximum for lysimeter-L and minimum for lysimeter-LSDL (Table 3). Figure 10 indicates that the alkalinity value was increased cruelly over a short time at the initial stage up to 47 days for all lysimeters, from 358-8045, 305-7870, 392-8935 and 700-10070 mg/L for lysimeter-LSDL, LSTL, LC and L, respectively. Soon after that, due to high amount of leachate generation the alkalinity may be watered down. Alkalinity load was also found highest in open dump lysimeter and lowest in lysimeter using sandy loam soil as top cover (Table 3). From the above explanations, alkalinity from landfill leachate can be significantly deducted by operating a landfill as sanitary landfill with a sandy loam soil as top cover. Findings are valid according to Karnchanawong et al. (2009).

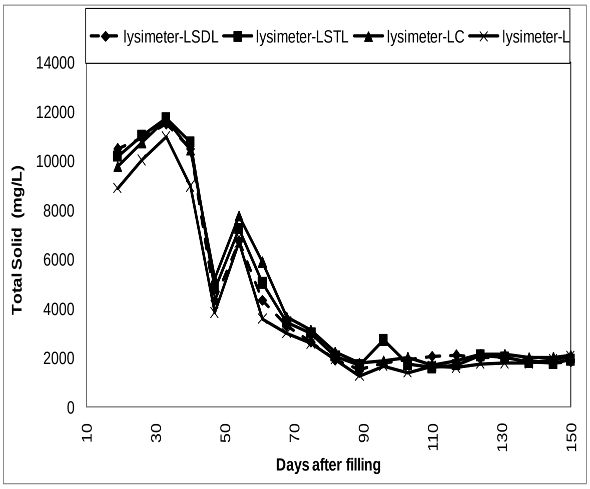

Figure 9 Variation of total solid in landfill lysimeter at varying operational condition

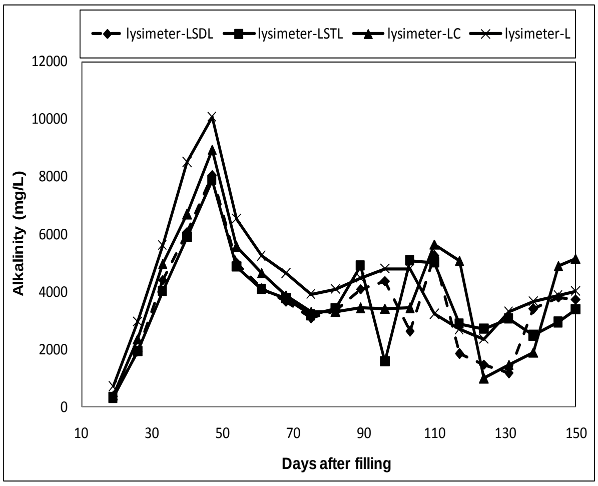

Figure 10 Variation of alkalinity in landfill lysimeterat varying operational condition

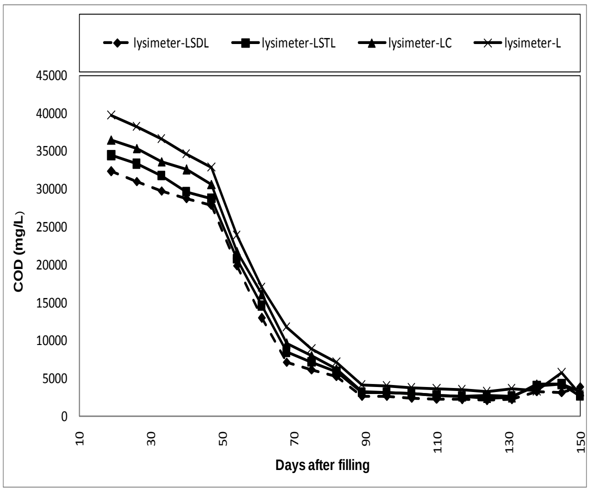

Figure 11 Variation of COD in landfill lysimeterat varying operational condition

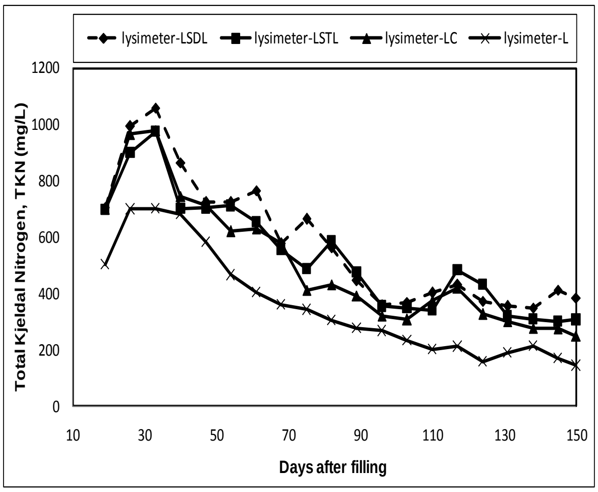

Figure 12 Variation of TKN in landfill lysimeter at varying operational condition.

COD Concentration

COD is an important test and it gives a quick measurement of pollution load of the leachate. Tables 3 and 4 reflect that the lysimeter-L posses the utmost average concentration and cumulative amount of COD load while lysimeter-LSDL posses the slightest. COD concentration was found highest for the first sampling from all the concerned lysimeters and Subsequently, the graph was declining in nature from 32305-2065, 34460-2580, 36505-2350 and 39725-2800 mg/L for lysimeter-LSDL, LSTL, LC and L, respectively, from the Figure 11. COD decreases sharply between 50-90 days due to comparatively the higher rate of leachate production (Rafizul et al. 2012). Therefore it is proved that an open dump landfill operational mode produced the most tainted leachate where sandy loam soil has a great efficiency to remove COD. The findings are supported by (Rafizul et al. 2012).

Total Kjeldahl Nitrogen

Nitrogen which has potential to pollute water and soil is another major constituent in the leachate. Total Kjeldahl Nitrogen (TKN) is the sum of organic nitrogen (organic-N) and NH_4-N (APHA 1998). The NH_4-N represents the inorganic compounds of leachate. Figure 12 depicts that TKN concentration of different landfill lysimeter operational mode varies from 1057.2 – 355, 975.3 - 301.7, 978.7 – 250 and 702.3 - 145.7 mg/L for lysimeter-LSDL, LSTL, LC and L, respectively. Concentration values were decreasing in nature with the elapsed period of experimental operation. Concentration of NH_4-N decrease with time with the increase of the age of landfill due to hydrolysis and fermentation of nitrogenous fractions of biodegradable refuses substances (Abbas et al. 2009). So the results are fairly correct according to Abbas et al. (2009). TKN concentration as well as load during the experimental period was found as highest in lysimeter having sandy loam soil as top cover and lysimeter–L produced the least. Rafizul et al. (2012) and Tubtimthi (2003) have implemented and experimentally investigated exactly the same results. Hence, it can be concluded that open dumping mode of landfill operation has a great efficiency in reduction of TKN rather than sanitary operational mode.

CONCLUSIONS

The methodical supervising of physico-chemical characteristics of leachate from MSW deposited in lysimeter has revealed that the comprehensive field level landfill simulation with varying operational condition by using different soil as top cover affected the quality of leachate vastly. Result reveals that open dump lysimeter produced more quantity of leachate around 24 % in contrast to the sanitary landfills. In addition, waste settlement rate in open dump lysimeter is the maximum contrary to the sanitary landfill lysimeters. Though open dump landfill operation has proved to be beneficial for reducing nitrogen concentration but rest constituent’s load in leachate were found as highest compared to sanitary landfills. Among all the lysimeters using cover soils, the lysimeter having sandy loam soil as top cover had the lowest concentration and load of most of the leachate constituents and least amount of leachate as well, against the other counterparts i.e the lysimeters having silty loam and clay soil.

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Investigation of Leachate Concentration in Landfill And Reduction by Different Coagulants

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ABSTRACT

Coagulation–flocculation is a relatively simple physico-chemical technique of leachate which has been practiced using a variety of conventional chemical coagulants. In this study, performance of coagulation and flocculation process was evaluated for leachate from solid waste disposal site at Rajbandh, Khulna city, Bangladesh. Leachate samples were collected periodically and investigated in the laboratory in terms of organic and inorganic compounds, metal and heavy metal concentrations. In this study poly-aluminum chloride (PAC) and mixture of Alum ($Al_2 (SO_4)_3 . 18H_2O$) *and Ferric chloride* ($FeCl_3.6H_2O$) *were used as coagulant. Effects of chemical and physical variables (concentration, pH, speed and time of rapid mixing and slow mixing) were examined by Jar-test experiments. The optimum pH and dosage of PAC were obtained as 7.6 and 1.5g/L respectively. The most efficient rapid and slow mixing speeds were 40 rpm for 1 minute and 30 rpm for 15 minutes, respectively. The optimum settling time was 30 minutes. Results showed that reduction of COD, SS, color and turbidity were 63%, 88%, 99% and 99% respectively for rapid mixing and 71%, 94%, 99% and 96%, respectively, for slow mixing.*

INTRODUCTION

Sanitary landfilling plays an important role in solid waste management in developed and least developed countries in the world. Moreover, landfill leachate is a widely used term in the environmental sciences where it has the specific meaning of a liquid that has dissolved or entrained environmentally harmful substances which may then enter into the environment. It is a heavily polluted wastewater which has high potential to pollute subsoil and ground water. Municipal solid waste (MSW) generates due to disposal of waste materials from domestic, commercial, institutional, market and other sources which might contain hazardous substances. It is reported that about 90% of municipal solid waste (MSW) is disposed in open dumps and landfills unscientifically, creating problems to public health and the environment [1]. Landfilling is one of the most popular methods of municipal solid waste (MSW) disposal due to its relative simplicity in terms of disposal procedures, low cost and landscape-restoring effect on the holes from mineral workings [2]. The generation of leachate is a significant concern associated to this disposal method. Leachate is a very dark colored, highly complex and polluted wastewater that is primarily formed by the percolation of precipitation through open landfill or through the cap of the completed site [3].

MSW leachate characteristics vary with time and from site to site because it depend on type of wastes disposed, rainfall, age of the landfill and design of the landfill etc [4]. Leachate can be categorized a liquid waste inheriting high chemical oxygen demand (COD), high biochemical oxygen demand (BOD), total suspended solid (TSS) , elevated values of total dissolved solid (TDS), Color, Turbidity, significant concentration of heavy metal and inorganic salts. [5].

It may also exhibit a wide variety of toxic and polluting components. In order to reach environment-friendly criteria for landfill leachate, one must bring these values to an acceptable discharge limit. Treatment and safe disposal of landfill leachate is essential as it could be a potential source of surface and ground water contamination and threaten the surrounding ecosystem. Therefore it is essential to collect the leachate emanating from the mass of waste and treat it before discharge to a sewer.

There are some wastewater treatment processes which are generally applied to treat landfill leachate: aerobic and anaerobic biological degradation, chemical oxidation, chemical precipitation, coagulation-flocculation, activated carbon adsorption and membrane processes. Physio-chemical processes are used for the pretreatment of young leachate to make it amenable to biological treatment, and to hydrolyze some refractory organic compounds found in leachate from older landfills. Biological treatment is primarily used to stabilize degradable organic matter in the young and middle aged leachates. In practice, a combination of physical, chemical and biological methods are usually used for the effective treatment of landfill leachate since it is difficult to obtain satisfactory effluent water quality by using any one of these methods alone [6].

Coagulation-flocculation is mainly used for treating stabilized stage and old age leachate [7]. It have many factors can influence the efficiency, such as the type and dosage of coagulant/flocculant, pH, mixing speed and time, temperature and retention time. The optimization of these factors may influence the process efficiency. The coagulation process destabilizes colloidal particles by the addition of a coagulant. To increase the particles size, coagulation is usually followed by flocculation of the unstable particle into bulky floccules so that they can settle more easily. The general approach for this technique includes pH adjustment and involves the addition of ferric/alum salts as the coagulant to overcome the repulsive forces between the particles [8].

This experimental study was conducted to investigate the effect of treatment of landfill leachate by a coagulation-flocculation process. The effects of different dosages of coagulant and different pH values on the coagulation processes were also compared. Poly aluminium chloride (PAC) and the mixture of alum with ferric chloride were tested as conventional coagulants. This study also aimed to establish the removal pattern of color in relation to the organic matter removal (COD) and associated suspended solids and turbidity.

MATERIALS AND METHODS

Sampling and Leachate Characterization

This research was focused on the performance of coagulation and flocculation process on stabilized leachate generated from solid waste disposal site at Rajbandh, situated in Khulna, Bangladesh. This leachate sample from MSW, deposited in Rajbandh, was collected periodically and has been investigated in the laboratory.

The samples were collected from five samplings at one month interval for about 5 months from 1st January to 28th May, 2012. After each sampling leachate characteristics were examined and the average value of five samplings was shown in Table 1.

Table 1 Characteristics of Raw Leachate used in this study

Characteristics	**Value**
Chemical oxygen demand (COD)	3600
Suspended solid (SS)	530
Color	9540
Turbidity	831
pH	8.76
Chromium (Cr)	14
Lead (Pb)	0.45
Zinc (Zn)	12
Nickel (Ni)	0.04
Copper (Cu)	4
Total Iron (Fe)	4.2

All values in mg/L, except pH, turbidity (NTU) and leachate color (Pt-Co). Average of five leachate samples has been taken from 1st January to 28th May, 2012.

The leachate samples were taken from semi aerobic landfill site at Rajbandh, Khulna and filled in 18 liters of plastic container at each time, transported to the laboratory. Then the samples were stored at about 28^0C for conditioning. For re-suspension of possible settling solids, samples had agitated thoroughly, before any test was conducted. The samples were analyzed for temperature, pH, COD, suspended solids (SS), color, turbidity and alkalinity in accordance with the Standard Methods for the Examination of Water and Wastewater.

Analytical Methods for the Assessment of Leachate

In the laboratory, pH was determined by pH meter (HACH, Model No. Sens ion 156) and conductivity by conductivity meter (HACH, Model No. Sens ion 5). Apparent color and COD were measured by a HACH DR/120 portable datalogging spectrophotometer, (reported in units of Pt/Co and mg/l, programs 120 and 435, respectively). Moreover, chloride was determined by potentiometric titration method using silver nitrate solution, alkalinity by titration method, hardness by EDTA titrimetric method as well as COD by closed reflexive method as per the Standard Methods (APHA 1998). In addition, total solid (TS) dried at 103-105^{0}C, total dissolved solid (TDS) dried at 180^{o}C and total suspended solid (TSS) dried at 103-105^{o}C were determined in the laboratory. In addition, Heavy metals viz., Cu, Cr, Cd, Ni, Pb, Mn, Fe and Zn were analyzed using spectrophotometer (HACH; DR/2400) in accordance with the Standard Methods for the Examination of Water and Wastewater [9].

Coagulation-flocculation Experiments

In this study, coagulation-flocculation process was performed in usual jar test equipment. The conventional jar test apparatus, equipped with 6 beakers of 1L volume, was employed for coagulation-flocculation. Chemicals reagents used as coagulants included poly-aluminum chloride, alum ($Al_2 (SO_4)_3 . 18H_2O$) and ferric chloride ($FeCl_3.6H_2O$).

For re-suspension of possibly settling solids leachate samples were thoroughly shaken and the appropriate volume of sample was transferred to the corresponding jar test beakers. A jar test was set up at room temperature for each trial. The coagulant was added into the beakers and the pH values were immediately adjusted to the desired levels by the addition of appropriate amounts of NaOH and HCl solutions.

The jar-test process consists of three distinct steps: (1) Rapid mixing stage; aiming to obtain complete mixing of the coagulant with the leachate to maximize the effectiveness of the destabilization of colloidal particles and to initiate coagulation. (2) Slow mixing stage; the suspension is slowly stirred to increase contact between coagulating particles and to facilitate the development of large flocs. (3) Settling stage; mixing is terminated and the flocs are allowed to settle [6]. To optimize the variables including pH, coagulant dose and speed & duration of rapid & slow mixing, respectively jar-test was used. These variables were optimized based on the highest percentage removal of the leachate constituents. Based on previous findings, the coagulant dose and pH for rapid and slow mixing were investigated and the initial speed and duration were assumed, presented in Table 2.

Table 2 Ranges of critical parameters obtained from other's findings

Critical parameter Range	References
Speed of Rapid Mixing (rpm)	20-100
Duration of Rapid Mixing (min)	1
Speed of Slow Mixing (rpm)	10-50
Duration of Slow Mixing (min)	15
Settling Time (min)	30-60

In order to determine coagulant dosage (optimum dose) on removal efficiency, different concentrations at stable pH such as 0, 0.5, 1, 1.5, 2, 2.5, 3, 3.5 mgL^{-1} of PAC and 0, 5, 1.2, 1.7, 2, 2.5 and 3 g/L for combination of alum and ferric chloride were added to 1 L leachate sample. After the settling period, the supernatant was digested using standard methods to release its organic matter and heavy metal contents and analysis were carried.

RESULTS AND DISCUSSION

Optimal Coagulant Dose (OpCD)

Coagulant dosage plays an important role in the removal of target pollutants in coagulation-flocculation studies. The optimum dosage of coagulant is defined as a value above which there is no significant increase in removal efficiency with further addition of the coagulant [10]. PAC and the combination of alum with $FeCl_3$ were used as coagulant and different doses were tested at neutral pH. Figure 1 and 2 represent the results of the experiments. The experimental data shows that the OpCD were 1.5 g/L for PAC and 1.2 g/L for alum with $FeCl_3$. The percentage of removal increases with an increase in the dosage up to the OpCD.

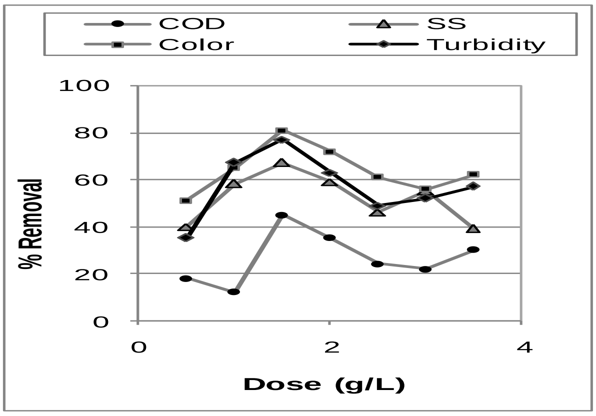

Figure 1 Percentage reduction of COD, SS, color, and turbidity using PAC dose at neutral pH.

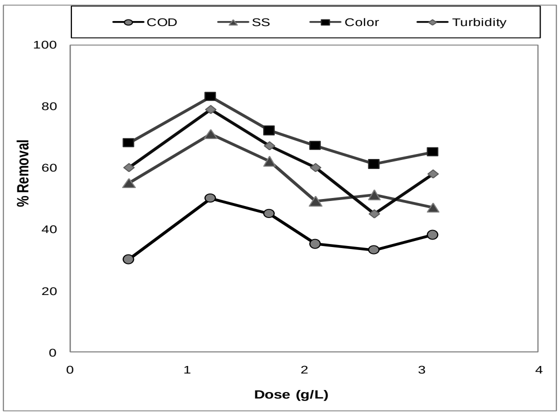

Figure 2 Percentage reduction of COD, SS, color and turbidity using Alum and $FeCl_3$ dose at neutral pH.

The values of percentage reduction of COD, SS, color and turbidity resulting from the OpCD at neutral pH were 45, 67, 81, and 77% respectively for PAC and 50, 71, 83 and 79% respectively for the combination of alum with $FeCl_3$. With increasing coagulant dose, the supernatant pH decreased gradually. The pH of the supernatant was 7.6 and 7.86 with OpCD of PAC and the combination of alum with $FeCl_3$. For the different coagulant doses tested, the flocs were microscopic and of relatively the same size. The sludge volume increased gradually with increasing coagulant dose up to OpCD and decreased thereafter.

p^H Influence

The coagulation pH is the most important parameter in the leachates coagulation process [11]. To perform the coagulation-flocculation tests, different pH values were tested while the OpCD was applied. Figures 3 and 4 show the range of pH examined was from 2 to 12, while 8.64 was the original pH value of raw leachate. When the range of pH was 7.5 to 9, efficiency of coagulation-flocculation was obtained From Fig. 3 and 4. For PAC and the alum with $FeCl_3$, the optimum pH was 7.6 and 7.86 respectively. The percentage reduction of COD, SS, color and turbidity resulting from the OpCD at pH 7.6 were 69, 87, 100, and 99.7% respectively for PAC and the values were 100, 87.04, 98.46 and 99.08% respectively for Alum with FeCl3. Some flocs were appeared without adding coagulant after pH adjustment from pH 2 to 12.

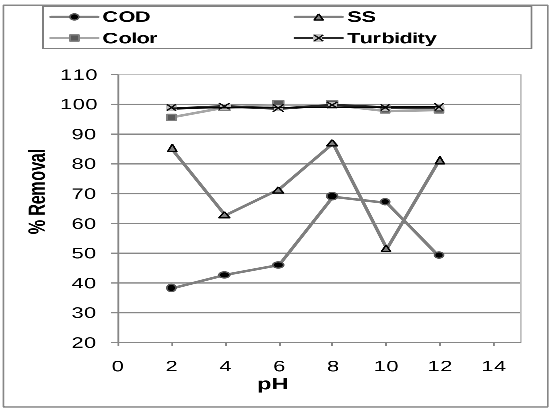

Figure 3 Percentage reduction of COD, SS, color and turbidity at varying pH using PAC at OpCD

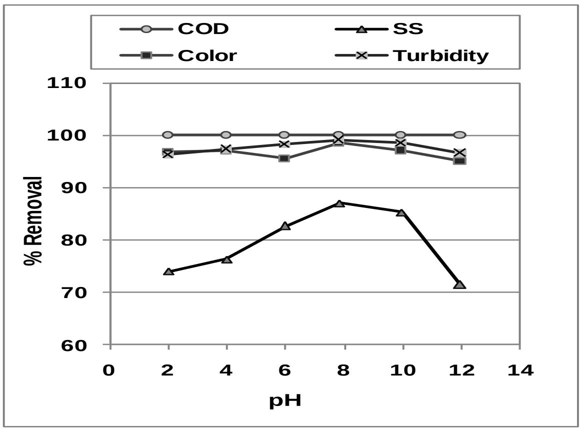

Fig.4 Percentage Reduction of COD, SS, color and turbidity at varying pH using Alum and $FeCl_3$ at OpCD.

Effect of Rapid Mixing

For Rapid mixing, the speeds were varies between 20 and 100 rpm with a duration of 60 seconds. OpCD and optimum pH were used for the experiments. The initial speed and duration of rapid mixing were 20 rpm and 60 seconds, to improve the initial values an investigation was made to examine the effect of different speeds (20, 40, 60, 80 and 100 rpm) during 60 seconds, then the efficient speed was determined. The best speeds were obtained as 40 rpm and 60 rpm for PAC and alum with $FeCl_3$

respectively. By using PAC and alum with $FeCl_3$ the percentage reduction of COD, SS, color and turbidity were 63, 87, 98, 99% and 100, 95, 97, 99% respectively shown in Figure 5 and 6.

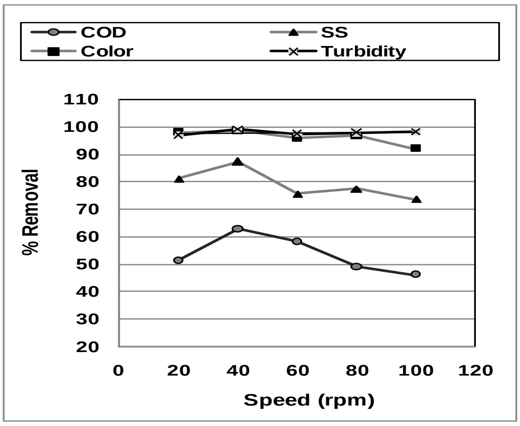

Figure 5 Percentage reduction of COD, SS, color and turbidity at speed of rapid mixing for 60 second using PAC

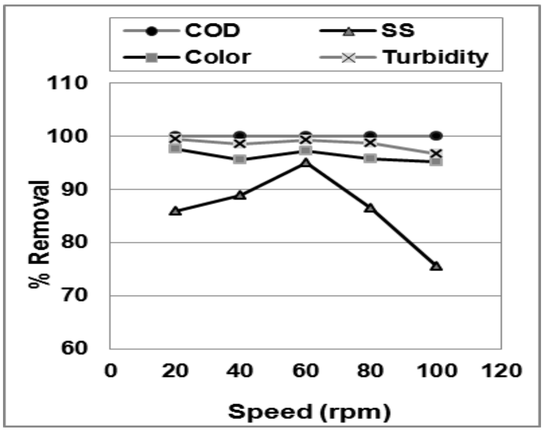

Figure 6 Percentage reduction of COD, SS, color and turbidity at speed of rapid mixing for 60 second using Alum and $FeCl_3$

Effect of Speed and Duration of Slow Mixing

Speed variation

Slow mixing was conducted to facilitate the best contact between coagulating particles. To investigate the appropriate speed of slow mixing where speed range of 10 to 60 rpm and mixing duration speed 15 minutes were examined. Figure 7 and 8 present that 15 rpm and 30 rpm were the most efficient speeds in the examination of different speeds for 15 minutes for PAC and alum with $FeCl_3$ respectively. Figure 7 presents the highest percentage of removal in COD, SS, colour, and turbidity were 71, 93.5, 98.79, and 96.41% respectively for PAC and Where figure 8 presents the values of the above parameters 100, 98.11, 98.9 and 97.82% respectively for alum with $FeCl_3$. In coagulation -flocculation process, normally the settling time was about 30 minutes (Baeza et al., 2004). For this experiment, settling time was about 60 minutes.

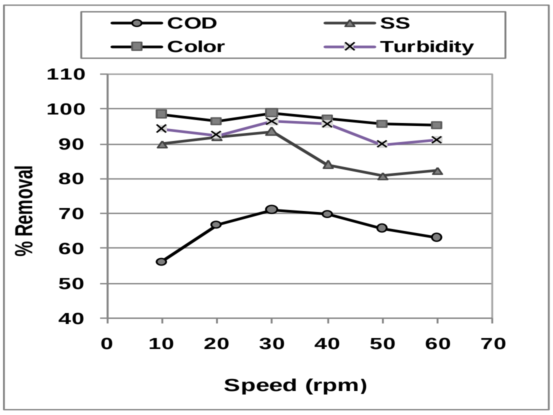

Figure 7 Percentage Reduction of COD, SS, color and turbidity at speed of slow mixing for 15 minutes using PAC

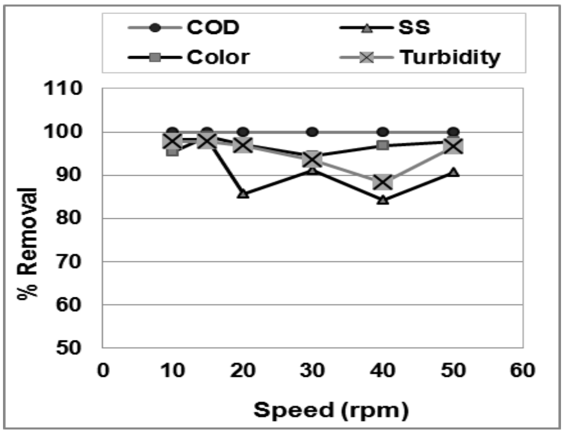

Figure 8 Percentage Reduction of COD, SS, color and turbidity at speed of slow mixing for 15 minutes using Alum and $FeCl_3$

Duration variation

Then different durations of slow mixing applying the speed of 30 rpm for PAC and 15 rpm for the combination of alum with $FeCl_3$ were tried. Figure 9 and 10 presents the effectiveness of slow mixing duration. The results show that the best parameter removal was achieved at the speed of 30 rpm for 15 minutes mixing for PAC and 15 rpm for 15 minutes mixing for the combination of alum with $FeCl_3$. The highest percentage of removal in COD, SS, colour, turbidity are 69.31%, 96.34%, 97.5% , 96.62% respectively for PAC and 100, 89.63, 98.32, 99.63% respectively for alum with $FeCl_3$ in the

Figure 9 and 10. The percentage for PAC and alum with $FeCl_3$ were in the same settling time which was 30 minutes.

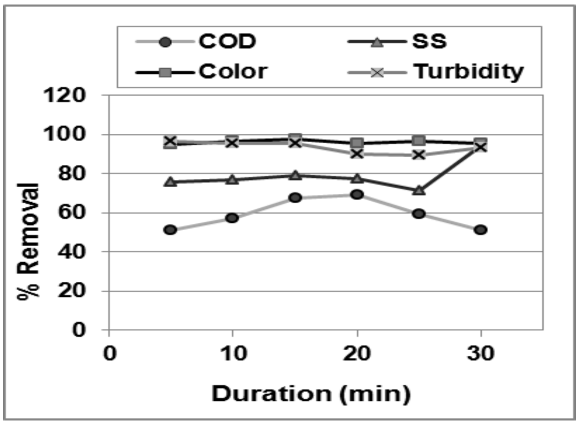

Figure 9 Percentage reduction of COD, SS, color, and turbidity vs. duration of slow mixing at the speed of 40 rpm using PAC

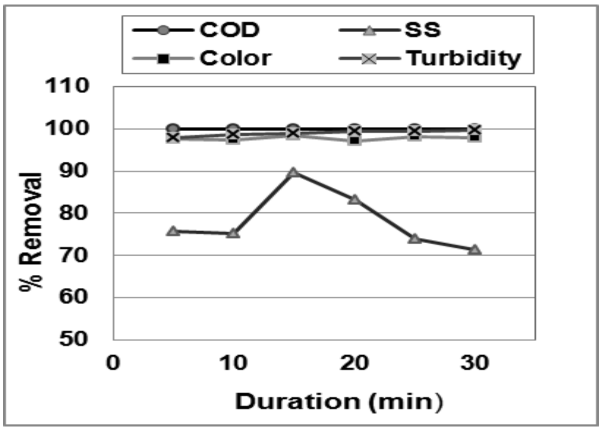

Figure 10 Percentage reduction of COD, SS, color, and turbidity with duration of slow mixing at the speed of 15 rpm using Alum and $FeCl_3$

4.4 Effect of Coagulation Process on Metal and Heavy Metal Concentration

The concentrations of metal and heavy metal in leachate ware investigated in the laboratory and hence presented in Table 3.

Table 3 Characteristics of Heavy Metal before and after treatment

Nitrogen, metal and heavy metal concentration	Initial Value of leachate (mg/L)	Percentage removal of heavy metals by using different coagulants	
		PAC	Alum with $FeCl_3$
Sodium (Na)	950	98	94
Potassium (K)	720	70	74
Calcium (Ca)	240	31	27
Magnesium (Mg)	130	62	76
Total kjeldahl nitrogen (TKN)	100	40	45
Ammonia nitrogen (NH_3-N)	47	51	46
Chromium (Cr)	19	74	90
Lead (Pb)	5	76	75
Zinc (Zn)	5.1	63	69
Nickel (Ni)	4	68	55
Copper (Cu)	37	46	49
Total Iron (Fe)	5	60	68

CONCLUSIONS

Results showed that PAC was categorized as good efficiency in leachate treatment. But the mixture of alum with ferric chloride was more effective in leachate treatment compared to PAC. However, PAC was achieved higher percentage removal in color. The experimental study showed that the OpCD were 1.5 g/L for PAC and 1.2 g/L for alum with $FeCl_3$. COD, SS, color and turbidity removal as a function of PAC and alum with ferric chloride dosage is shown at different pH values (from pH 2 to pH 12). The highest percentage of removal in COD, SS, color and turbidity were 71%, 97%, 100% and 100% for PAC and 100%, 98%, 99% and 100% for alum with ferric chloride. Alum and ferric chloride provides the highest percentage of removal in COD, SS, color and turbidity compared with PAC.

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Solar Disinfection Treatment of Surface Water

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ABSTRACT

Surface water is one of the main sources of fresh water in Bangladesh. Although ground water is available throughout the country, high concentration of arsenic, iron and salinity in ground water, leading the use of surface water instead of ground water. The contamination of surface water by faecal coliform necessitates its treatment. As a low cost household technology the effectiveness of Solar Disinfection method to treat the contaminated water is recognized throughout the world. The present study aims at resolve of the effectiveness of this method in Bangladesh. To achieve the goal the effectiveness of Solar Disinfection method in different climatic conditions of Bangladesh is evaluated. The study revealed that the method is very effective in both cold (December) and hot (April) weather. In addition, the effects of bottle qualities on Solar Disinfection method were also evaluated. The increase or decrease of pH than 7.3 increases the inactivation rate of bacteria. The poly ethylene terephthalate (PET) bottle showed higher inactivation rate of microorganisms than glass bottle. The bottle size ranges from 0.5L to 1.5L have no significant influence on the inactivation rate of Total coliform (TC) and Escherichia coli (EC). Another test was done to evaluate the difference of inactivation kinetics of bacteria in a cloudy day and sunny day. The test showed that the TC was not fully removed in the cloudy day while in the sunny day total removal was observed. All the EC were removed within 5 hour both in cloudy day and sunny day i.e., the sunlight does not play a major significant role on the inactivation rate of EC. However, the study proves that the Solar Disinfection method can contribute to the treatment of water in household level.

INTRODUCTION

Since times immemorial surface water has been using for drinking purpose in Bangladesh. The introduction of tubewell reduced the plentiful use of surface water. Tubewell turned the use of surface water into groundwater. Though groundwater fulfills the necessity of potable water in almost every part of Bangladesh, recently a lot of problems like excessive concentration of arsenic, iron and salinity in groundwater and lowering of ground water level have created obstructions to the use of the groundwater which lead again to the use of the surface water (Ahmed and Rahman 2000, Net1). In Bangladesh, rainwater is also used with limit as another alternative source of both surface water and groundwater. Different study reveals that both surface water and rainwater are contaminated that necessitate treatment (Ahmed and Rahman 2000, Net2). For treating water a lot of techniques have been introduced. These techniques are definitely a burden to a group of people who lives below the poverty line and fail to bring their daily bread. However, the solar disinfection method may be an appropriate solution. It is reported that a large amount of microbes, including pathogenic bacteria and viruses, can be killed by exposing the contaminated water to sunlight. The methods include the placing of the water inside a transparent glass container or a transparent plastic container and place it in the direct sunlight for several hours before drinking. The method can play a great role in Bangladesh for the poor communities as the method need no significant financial investment. But sufficient researches have not yet been done in Bangladesh on this method. This insufficiency inspired this project to work on this method. The project includes the evaluation of the effectiveness of solar disinfection method in two climatic conditions of Bangladesh. The effects of water qualities and bottle characteristics, initial concentration of microorganisms and cloud on the sky were also investigated.

EXPERIMENTAL PROCEDURE

The test bottles were collected from a nearby local market beside KUET so that the commercially available bottles can be used in the research work. The characteristics of the bottles are shown in Table 1.

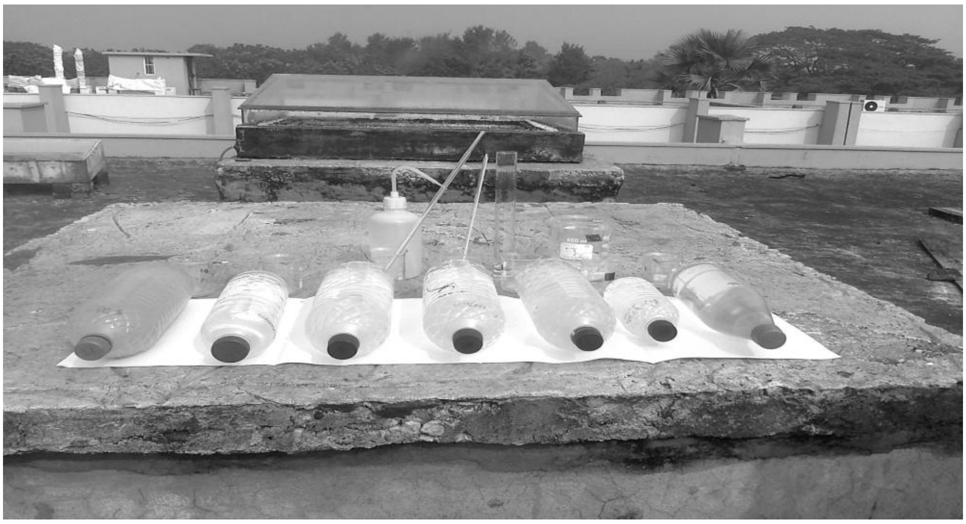
Table 1 Characteristics of the bottles used for the research**.**

Constituting Material	Volume of Bottle (Litre)	No. of Bottle	Diameter (mm)	Thickness(mm)
Polyethylene terephthalate	0.5	1	63	0.5
Polyethylene terephthalate	1	1	76	0.5
Polyethylene terephthalate	1.5	4	101	0.5
Glass	1	1	76	4

A suitable water source was selected considering the aesthetic attitude of mass people. All the tests for the research work were done in the Environment Laboratory of Civil Engineering Department of Khulna University of Engineering and Technology which is very near to the selected pond. For this reason the water was directly filled in the sterilized bottles and carried to the laboratory without any icebox. A sample was tested in the month of December. The amounts of Total coliform (TC) and Escherichia coli (EC) at each hour were recorded by membrane-filter method. The temperature of water was also recorded at each hour. In April the test was repeated for sample from the same source. The sample was also tested for different pH and turbidity in the same month. The pH was decreased using HCl (1M) and increased using NaOH (1M). Some sterilized soil was mixed to increase the turbidity. 0.5L, 1L and 1.5L PET bottle were used to determine the effect of bottle size on SODIS. The data of 1.5L PET bottle was used as the data of the normal water sample for evaluation of the effect of pH and turbidity. The test was also done in a 1L glass bottle to compare the data of the 1L PET bottle to determine the effect of bottle constituting material on this method. The test to evaluate the effect of characteristics of water and bottle was done in a cloudy (partial sunny) day so that the data of 1.5L PET bottle can be compared with the data founded from the test done on the full sunny day. The bottle/bottles was/were placed horizontally on the roof of the Civil Engineering Department of Khulna University of Engineering & Technology, Khulna, Bangladesh as shown in Figure 1. The Table 2 represents the date, time and No. of sample of the various tests done.

Table 2 Date, time and No. of sample of the various tests done

Date	Time	No. of Sample	Test Name
1st Dec., 2011	9.00 A.M. to 5.00 P.M.	01	Inactivation Kinetics of Bacteria.
1st April, 2012	9.00 A.M. to 5.00 P.M.	01	Inactivation Kinetics of Bacteria.
1st Dec., 2012	9.30 A.M. to 4.30 P.M.	07	Effect of the Water Quality and Bottle Characteristics on the Solar Disinfection Method.



Figigure1 Placement of Bottles under Sunlight

RESULTS AND DISCUSSION

The Inactivation Kinetics of the Microorganisms in Two Different Seasons of Bangladesh

The Variation of temperature in bottles in two different seasons December and April in Bangladesh is shown in Figure2. Both of the Figures 3 and 4 shows the inactivation rate of microorganisms. The rate in April is higher than December. The Figure 3 shows that the complete removal of TC was found in April, whereas incomplete removal was found in December that.

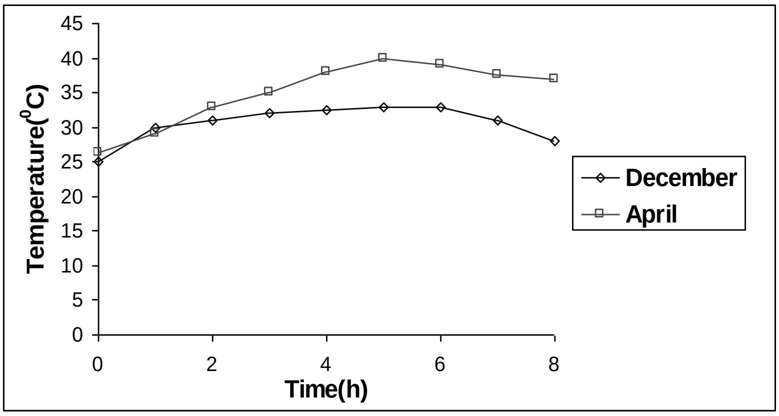


Figure 2 Variation of temperature in bottles in two different seasons

The Figure 2 indicates that there is notable difference between temperatures of the two months

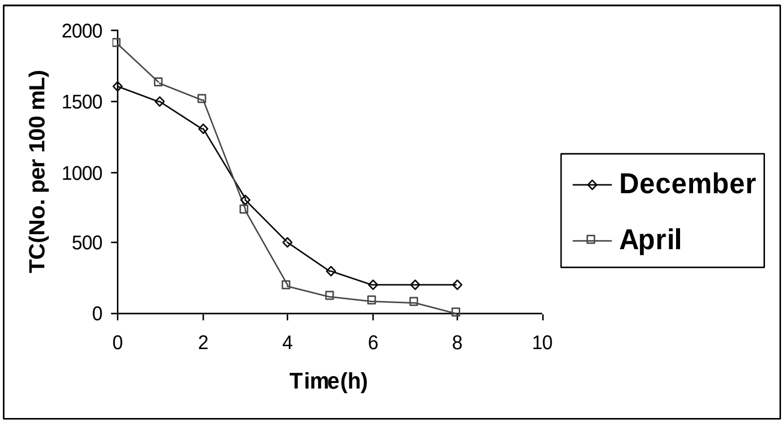


Figure 3 Inactivation kinetics of TC in two different seasons

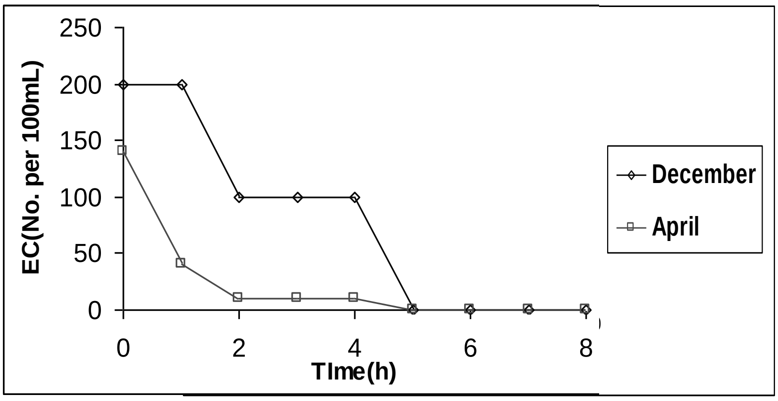


Figure 4 Inactivation kinetics of EC in two different seasons

The Effect of pH on Inactivation kinetics of Microorganisms

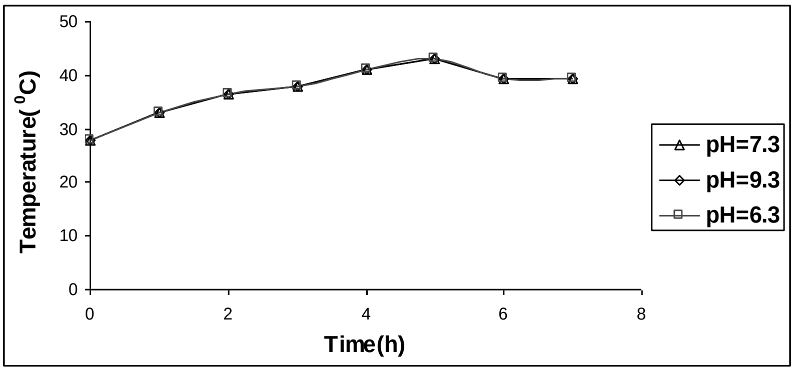


Figure 5 Variation of temperature due to variation of pH

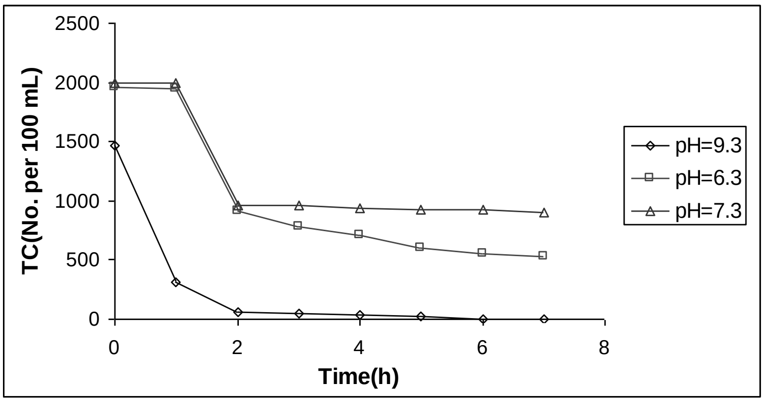


Figure 6 Inactivation kinetics of TC due to variation of pH

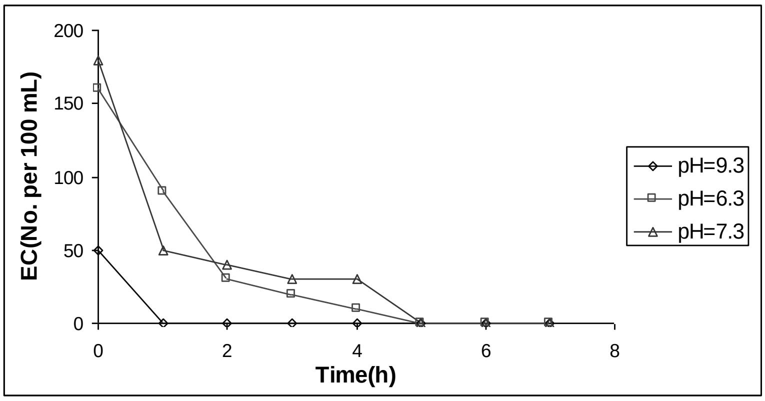


Figure 7 Inactivation kinetics of EC due to variation of pH

The figure 5 shows that due to change in pH of water the temperature of water does not change. The figure 6 shows that the increase or decrease in pH increased the Microorganisms' inactivation kinetics. The curve of water of pH=9.3 showed very steep slope from the beginning of test. The curve of water of pH=6.3 and pH=7.3 showed same inactivation rate for first 2 hours. But after 2 hour for water of pH=7.3 the inactivation rate became almost constant while the water of pH=6.3 shows continuous inactivation although the rate was less than first two hours. The figure 7 shows complete removal of EC with irregular inactivation kinetics. However, pH has significant role on bacterial existence (Net3).

4.3 The Effect of Turbidity on Inactivation Kinetics of Microorganisms.

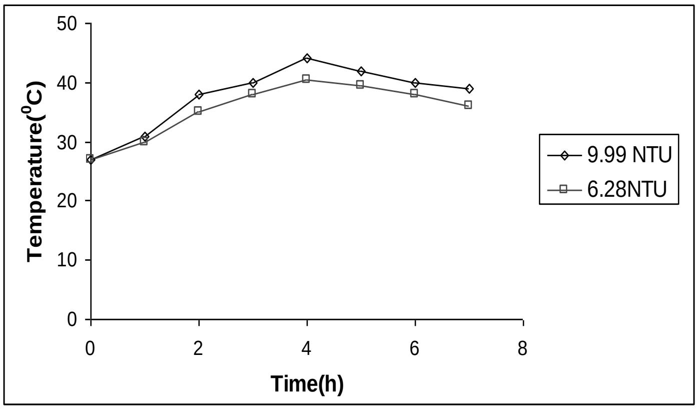


Figure 8 Variation of temperature due to variation of Turbidity

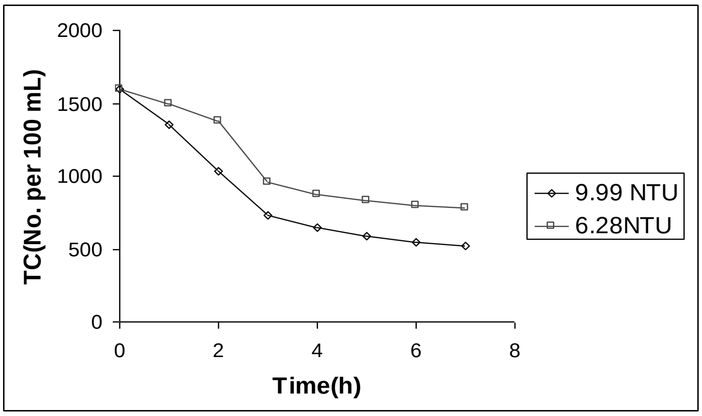


Figure 9 Variation of inactivation kinetics of TC due to variation of Turbidity

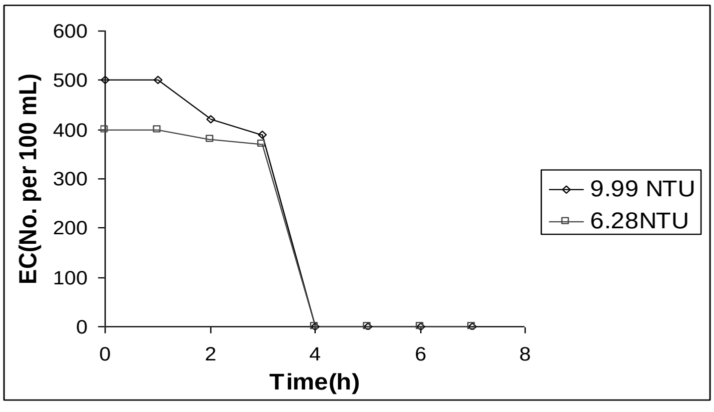


Figure 10 Variation of inactivation kinetics of EC due to variation of Turbidity

Figure 8 shows that the water temperature in slightly higher turbidity water was slightly higher than that of low turbidity water. Figure 9 and 10 show that inactivation rate of microorganisms in a slightly turbid water (turbidity 9.99 NTU) was slightly higher than that in a low turbidity water (turbidity 6.28 NTU).

4.4 The Effect of Bottle Constituting Material on Inactivation Kinetics of Microorganisms.

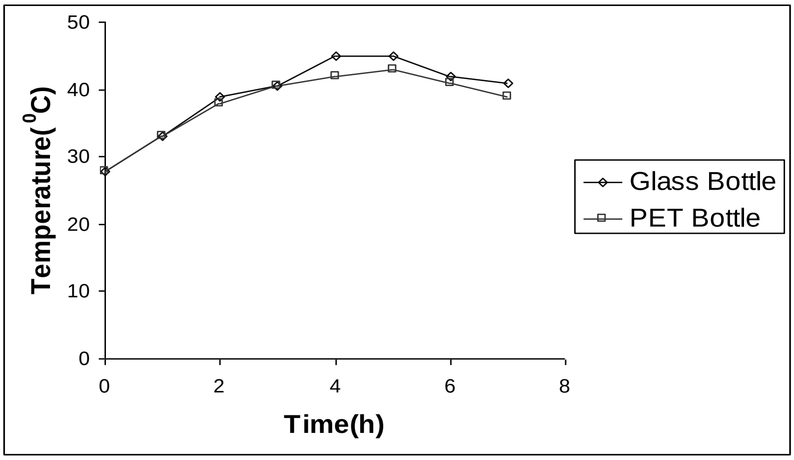


Figure 11 Variation of temperature due to variation of bottle constituting material

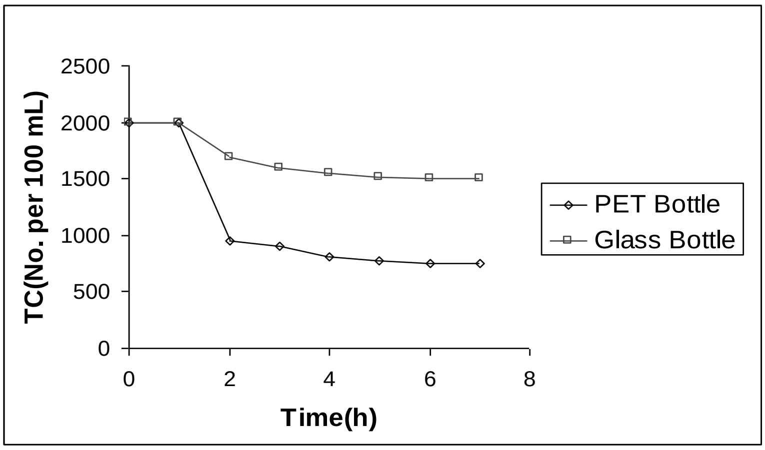


Figure 12 Inactivation kinetics of TC due to variation of bottle constituting material

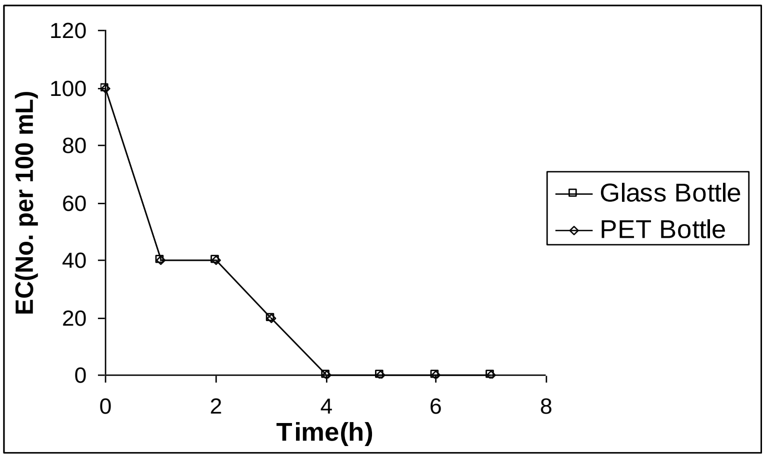


Figure 13 Inactivation kinetics of EC due to variation of bottle constituting material

The above figure 12 and figure 13 show that the inactivation rate of microorganisms is higher in PET bottle (thickness 0.5 mm) than a slightly greenish glass bottle (thickness 4 mm). But the figure 11 shows that the temperature of water at glass bottle is higher than water at PET bottle.

4.5 The Effect of Size of Bottle on Inactivation kinetics of Microorganisms.

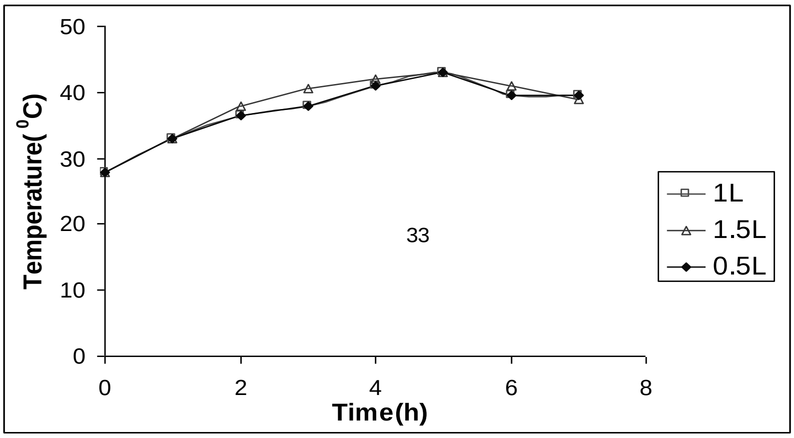


Figure 14 Variation of temperature due to variation of bottle size

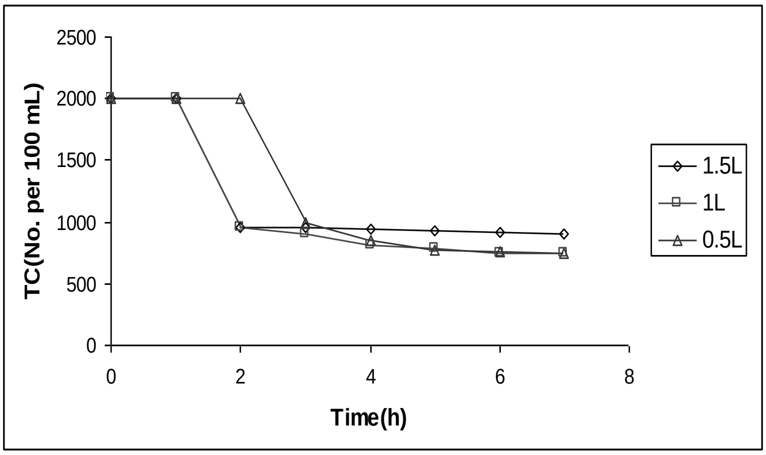


Figure 15 Inactivation kinetics of TC due to variation of bottle size

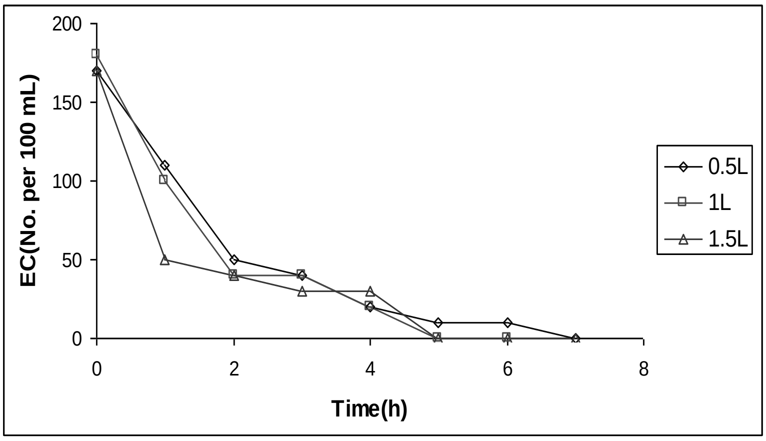


Figure 16 Inactivation kinetics of EC due to variation of bottle size

Water samples in 3 PET bottles of size 0.5, 1.0 and 1.5.0 L were exposed to sunlight for 7h. Figure 15 and 16 show that the bottle volume in the range of 0.5 to 1.5L does not significantly influence the reaction kinetics. This indicates that for household solar disinfection, bottles with a capacity up to 1.5 L can be used without adversely affecting the inactivation efficiency.

4.6 The Effect of Cloud on Sky on Inactivation Kinetics of Microorganisms.

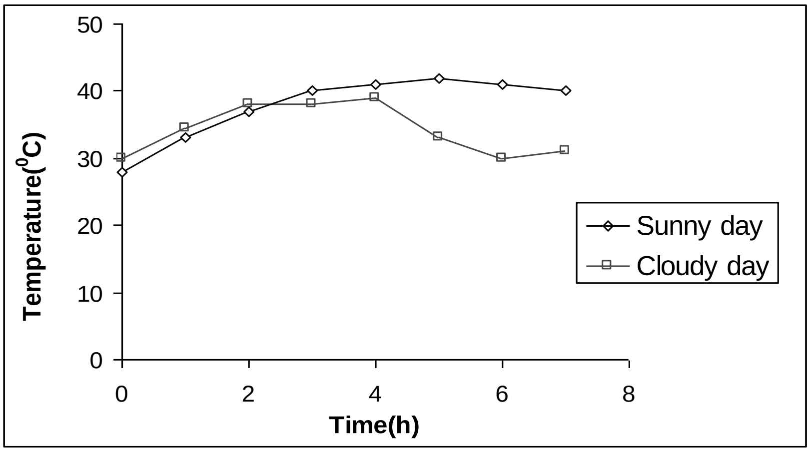


Figure 17 Variation of temperature in a cloudy day and sunny day respectively

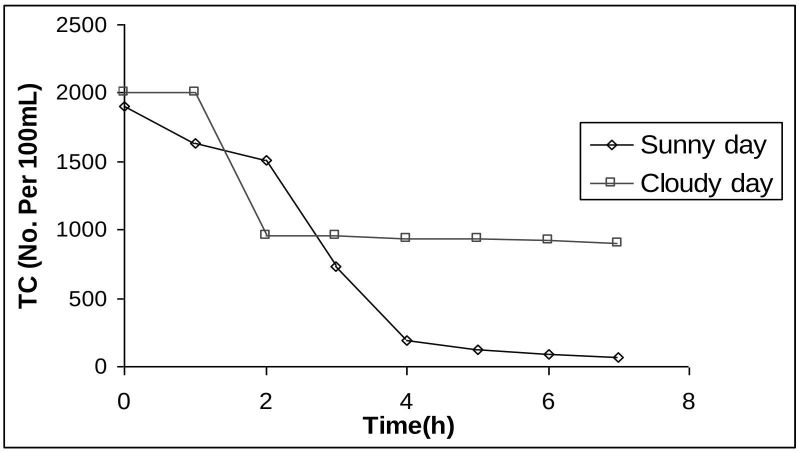


Figure 18 Inactivation kinetics of TC in a Cloudy Day and Sunny Day respectively

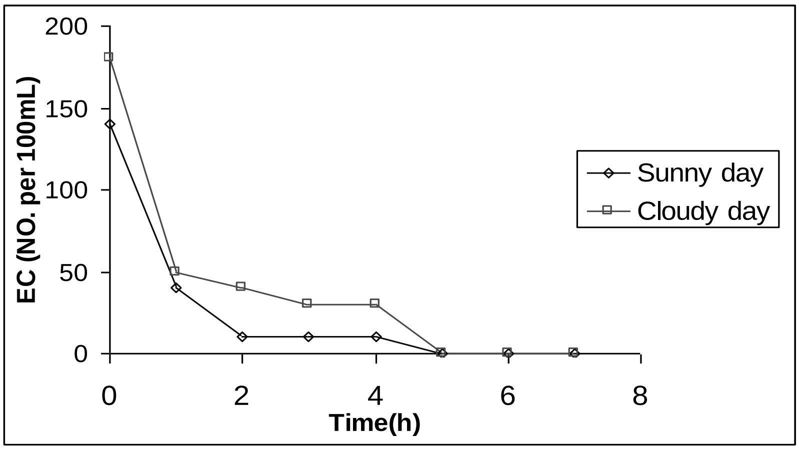


Figure 19 Inactivation kinetics of EC in a Cloudy Day and Sunny Day respectively

The figure 17 shows that the temperature in the cloudy day was higher than sunny day. The figure 18 is showing that the slope of inactivation curve of total coliform in a cloudy day is almost zero except the 2nd hour. The figure 19 shows that the total removal of E. coli for both sunny day and cloudy day took 5hour although the initial concentration of E. coli in sunny day was less than cloudy day. This implies that EC can be removed by Solar Disinfection in both sunny and cloudy day.

RECOMMENDATION FOR FUTURE RESEARCH

To propose a unique methodology that will ensure complete removal of microorganisms throughout the country some more factors are necessary to test as follows:

- In December complete removal of TC was not observed in a single day. So, whether the complete removal is possible or not possible in two days should be investigated.

- the effect of backing surface which will reflect or absorb Ultra-violet ray.

- the effect of angle of placement of bottle because it may change the intensity of sunlight.

CONCLUSIONS

On the basis of this study it can be concluded that:

- The method is effective in both cold (December) and hot (April) weather. Complete inactivation of EC is found in most of the cases with inactivation period mostly 4 to 6 hours. Mostly initial EC value 400 to 200 no/100 ml reduced to zero.
- The test showed that the TC was not fully removed in the cloudy day. All the EC were removed within 5 hour both in cloudy day and sunny day i.e., the sunlight does not play a major significant role on the inactivation rate of EC except day temperature. The increase or decrease of pH value of the raw sample water than 7.3 increase the inactivation rate.
- The poly ethylene tere-phthalate (PET) bottle showed higher inactivation rate of microorganisms than glass bottle. The bottle sizes range from 0.5L to 1.5L have no significant influence on the inactivation rate of TC and EC.

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Comparison of Different Chemical Coagulants for Treatment of Landfill Leachate

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ABSTRACT

Leachate migration from the landfill could be a potential source of surface and groundwater contaminations. Landfill leachate is a very dark colored liquid formed primarily by the percolation of precipitation through open landfill or through the cap of the completed site. A study was conducted to investigate the efficiency of coagulation and flocculation processes for removing color from municipal solid waste (MSW) deposited in Rajbandh at Khulna city, Bangladesh. Different types of coagulants such as poly aluminum chloride (PAC), ferrous sulpHate, mixture of ferrous sulfate with lime and mixture of alum with ferric chloride were studied using standard jar test apparatus. Then the results indicated that the maximum removal of color for the mixture of alum and ferric chloride was 98.4% at an optimum dose of 2.1 g/L at pH 4. And the removal was more by using PAC, which was 100% at an optimum dose of 1.7 g/L at pH 6.

INTRODUCTION

Landfills have been the most common methods of organized waste disposal in many places around the world. This method continues to be widely accepted and used due to such advantages such as simple disposal procedure, low cost, and landscape-restoring effect on holes from mineral workings. However, there is a significant concern for this method due to the production of a highly complex and polluted wastewater called leachate [1].

Leachate is a liquid generated as a result of rainwater percolation through wastes in a landfill [2]. Landfill leachate is a very dark colored liquid formed primarily by the percolation of precipitation through open landfill or through the cap of the completed site. The decomposition of organic matter such as humic acid may cause the water to be yellow, brown or black [3]. Leachates may contain high levels of organic matter including both biodegradable and non-biodegradable, suspended solids, inorganic salts and significant concentration of heavy metals [1]. It may also exhibit a wide variety of toxic and polluting components. In order to reach environment-friendly criteria for landfill leachate, one must bring these values to an acceptable discharge limit. Treatment and safe disposal of landfill leachate is essential as it could be a potential source of surface and ground water contamination and threaten the surrounding ecosystem. Therefore it is essential to collect the leachate emanating from the mass of waste and treat it before discharge to a sewer.

Leachate can be treated by three main methods that is pHysical, chemical and biological treatment. Treatment can be a combination of two or three of the above methods. Air stripping, adsorption are major pHysical leachate treatment methods, while the other methods such as coagulation-flocculation, chemical precipitation, chemical and electrochemical oxidation methods are the common chemical methods used for the landfill leachate treatment. This combination method is most popularly used to achieve excellent leachate treatment efficiency [4].The leachate treatment processes have different effectiveness depending on the leachate from landfill of different ages. The biological treatment process has been found to be effective on leachate from young landfills. Sometimes leachate can be treated by traditional package treatment plants on site, but this kind of treatment is the biological process, especially for young leachate [5].

Coagulation-flocculation is an effective method for the removal of non-biodegradable organic compounds and heavy metals from landfill leachate. The advantage of this method for the treatment of leachates are mainly simplicity, low cost, good removal efficiency and easy on site implementation [6]. The coagulation process destabilizes colloidal particles by the addition of a coagulant [2]. Ferric chloride, aluminium sulpHate and polychlorinated aluminium are commonly used as coagulant. Furthermore, polymer coagulant is used as auxiliary coagulant. The choice of coagulant chemical depends upon the nature of the parameter to be removed, the raw water conditions, the facility design, and the cost of the amount of chemical necessary to produce the desired result [7]. Coagulation-flocculation process is usually used for treating fresh leachate and it is applied as a pretreatment before biological treatment. It is used to remove heavy metal and non-biodegradable organic compounds from landfill leachate [8].

The main objective of this research was to investigate the efficiency of coagulation–flocculation process for the treatment of both fresh (raw), as well as of partially stabilized (by re-circulation) leachates. More specifically, the aim was the determination of most appropriate coagulant type and dose, the examination of pH effect on removal capacity, the investigation of combined action of coagulants–flocculants and the identification of optimum experimental conditions for the efficient application of this process. This research examined the effectiveness of PAC, alum, ferric chloride, ferrous sulfate and lime for the removal of color and heavy metal from semi-aerobic landfill leachate.

MATERIALS AND METHODS

Sampling and Leachate Characterization

The study focused on the leachate generated from Rajbandh at Khulna, Bangladesh. This site is subjected to highly colored and turbid leachate of 9540 PtCo due to the pressure of high organic matters associated with suspended solids and turbidity. This research was focused on the performance of coagulation and flocculation process on stabilized leachate generated from solid waste disposal site at Rajbandh, situated in Khulna city, Bangladesh. This leachate sample from MSW, deposited in Rajbandh, was collected periodically and has been investigated in the laboratory.

The samples were collected from five samplings at one month interval for about 5 months from 1st January to 28th May, 2012. After each sampling leachate characteristics were examined and the average value of five samplings was presented in Table 1.

Table 1 Characteristics of raw leachate used in this study

Characteristics	Value
Chemical oxygen demand (COD)	3600
Suspended solid (SS)	530
Color	9540
Turbidity	831
pH	8.76
Chromium (Cr)	14
Lead (Pb)	0.45
Zinc (Zn)	12
Nickel (Ni)	0.04
Copper (Cu)	4
Total Iron (Fe)	4.2

All values in mg/L, except pH, turbidity (NTU) and leachate color (Pt-Co)).Average of five samples taken from 1st January to 28th May, 2012

The leachate samples were taken from semi aerobic landfill site at Rajbandh, Khulna and filled in 18 liters of plastic container at each time, transported to the laboratory. Then the samples were stored at about 28^{0}C for conditioning. For re-suspension of possible settling solids, samples had agitated thoroughly, before any test was conducted. The samples were analyzed for temperature, pH, COD, suspended solids (SS), color and alkalinity in accordance with the Standard Methods for the Examination of Water and Wastewater [9].

Analytical Methods for the Assessment of Leachate

In the laboratory, pH was determined by pH meter (HACH, Model No. Sens ion 156) and conductivity by conductivity meter (HACH, Model No. Sens ion 5). Moreover, chloride was determined by potentiometric titration method using silver nitrate solution, alkalinity by titration method, hardness by EDTA titrimetric method as well as COD by closed reflexive method as per the Standard Methods (APHA 1998). In addition, total solid (TS) dried at 103-105^{0}C, total dissolved solid (TDS) dried at 180^{o}C and total suspended solid (TSS) dried at 103-105^{o}C were determined in the laboratory. In addition, Heavy metals viz., Cu, Cr, Cd, Ni, Pb, Mn, Fe and Zn were analyzed using spectropHotometer (HACH; DR/2400) in accordance with the Standard Methods for the Examination of Water and Wastewater (APHA,1998).

Coagulation studies on leachate were performed with jar test (Jar Test Model CZ150) equipment comprising six paddle roters, equipped with six beakers. Then different coagulants such as PAC, alum, ferric chloride, ferrous sulfate and lime were used with an optimum dose. The initial rapid mixing stage for both experiments were 1 min at 350 rpm, followed with slow mixing stage for 19 min at 50 rpm.The final gravity settling stage lasted for 1 hour.

RESULTS AND DISCUSSIONS

Effect of pH on Removal of Color

The coagulation pH is the most important parameter in the leachates coagulation process [10]. The influence of different dosages of coagulants at different pH values (pH 4, 6 and 12) for the removal of color was measured by using various coagulants (PAC, Alum, Ferric chloride, Ferrous sulfate and Lime).

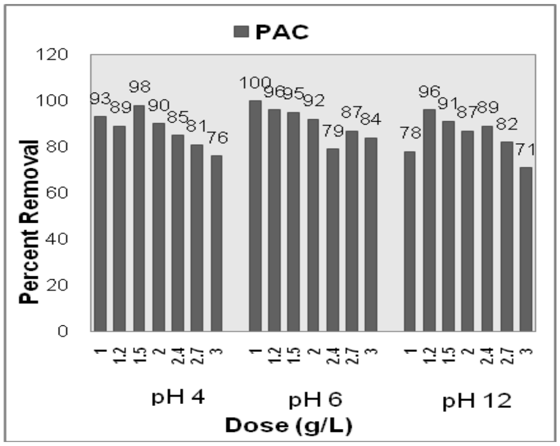

Figure 1 Percentage removal of color by using PAC doses with varying pH.

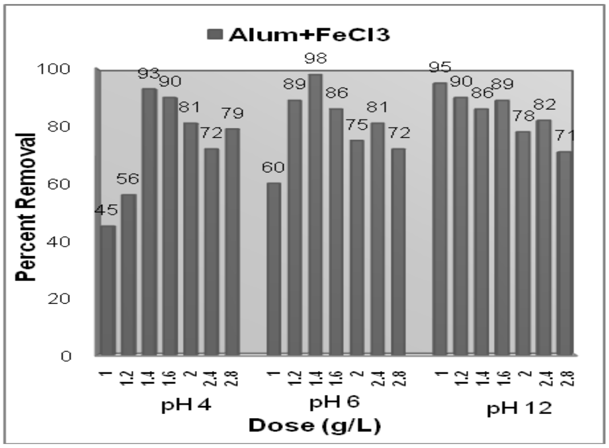

Figure 2 Percentage removal of color by using Alum with$FeCl_3$ doses with various pH

Figure 1 showed the percentage removal of color at different pH by using PAC as coagulant. At pH 4, 6 and 12, the maximum removal was found as 98%, 100% and 96% with an optimum dose of 1.5, 1 and 1.2 g/L respectively for PAC. In case of the combination of alum and $FeCl_3$, the percentage removal of color was found as 93%, 98% and 95% with an optimum dose of 1.4, 1.4 and 1 g/L at pH 4, 6 and 12 respectively shown in Figure 2. So, removal at pH 6 is better compare to pH 4 and pH 12.

From Figure 3 the percentage removal was found as 82%, 25% and 90% with an optimum dose of 2.4, 3 and 2.7 g/L at pH 4, 6 and 12 respectively by using combination of $FeSO_4$ & lime as coagulant. So, removal at pH 12 is better than others. Figure 4 presented that the highest percentage removal was 56%, 47% and 92% with an optimum dose of 3, 3.6 and 2.6 g/L at pH 4, 6 and 12 respectively by using $FeSO_4$ as coagulant. So, maximum removal was seen at pH 12.

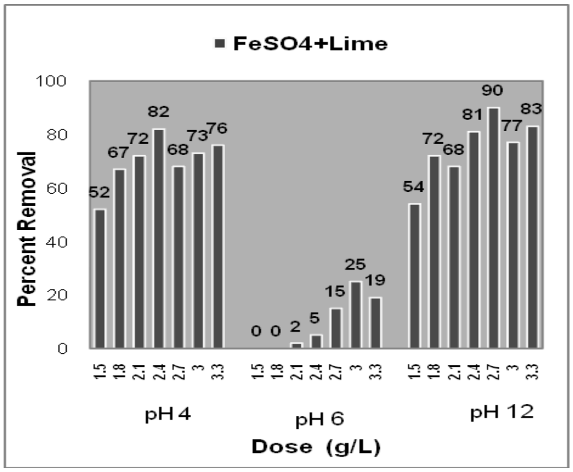

Figure 3 Percentage removal of color by using $FeSO_4$ with lime doses at varying pH

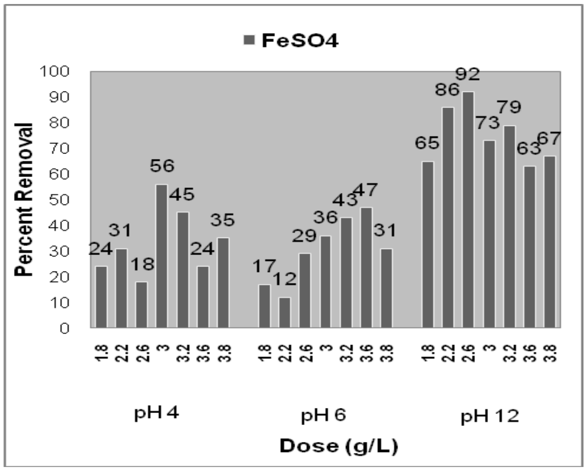

Figure 4 Percentage removal of color by using $FeSO_4$ at various doses with various pH

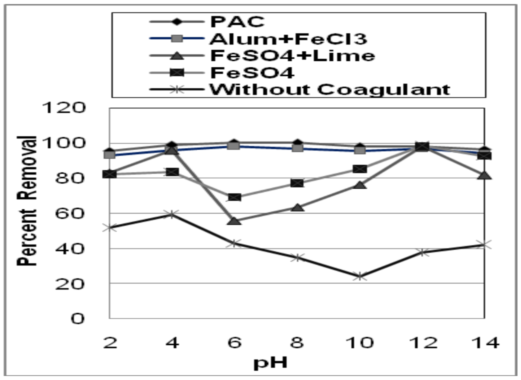

Figure 5 Pattern of color removal without and with adding optimum coagulant dose at different pH values

Results indicated that the color of leachate turned from black to light brown at lower pH to higher pH values. And the maximum removal was found as at 60% at pH 4 without using coagulant. PAC and alum with $FeCl_3$ exhibited the best results in terms of color removal. The higher percent of removal is 100% using PAC and alum with $FeCl_3$ at pH 6. In the case of $FeSO_4$ and lime, after pH 4 the removal started to decrease drastically with an increase in pH and reached the highest removal at pH 12.

Effect of Coagulant Dosage

Coagulant dosage plays an important role in the removal of target pollutants in coagulation-flocculation studies. The optimum dosage of coagulant is defined as a value above which there is no significant increase in removal efficiency with further addition of the coagulant [11]. Results showed that the color removal increased with an increase in coagulant dosage until it reached at an optimum value. This should be attributed by re-stabilization of colloidal particles when coagulants used at dosages in excess of the optimum value.

Table 2 Color removal at pH 4, 6 and 12 for various dosages of coagulants

pH	Results	PAC	Alum with $FeCl_3$	FeSO4 with Lime	$FeSO_4$
4	Optimum dosage (mg/L)	1500	1600	2400	2600
	Initial Concentration (PtCo)	3310	5560	1960	1856
	Final Concentration (PtCo)	65	385	406	1075
	% Color Removal	98	93	82	43
6	Optimum dosage (mg/L)	1000	1400	3000	1900
	Initial Concentration (PtCo)	4940	7560	3400	2730
	Final Concentration (PtCo)	0	201	2548	2185
	% Color Removal	100	97	25	20
12	Optimum dosage (mg/L)	1200	1000	2700	2300
	Initial Concentration (PtCo)	4540	3100	1916	1760
	Final Concentration (PtCo)	191	170	190	160
	% Color Removal	96	95	90	91

Effect of COD with respect to Color

The results of COD and color removal under the optimum conditions determined for each coagulant can be seen in Figure 5. The optimum results obtained from PAC, Alum with $FeCl_3$, $FeSO_4$ and $FeSO_4$ with Lime were 100% & 52%, 98% & 46%, 90% & 32% and 92% & 35% respectively for color and COD removal provided in Figure 6. According to the Color and COD results, it can be said that there is a direct relationship between the removal of these two parameters. The higher is the Color removal, the higher is the COD removal, but this only applies to the pHysio-chemical treatment of the leachate through the process of coagulation-flocculation-sedimentation with inorganic coagulants [10].

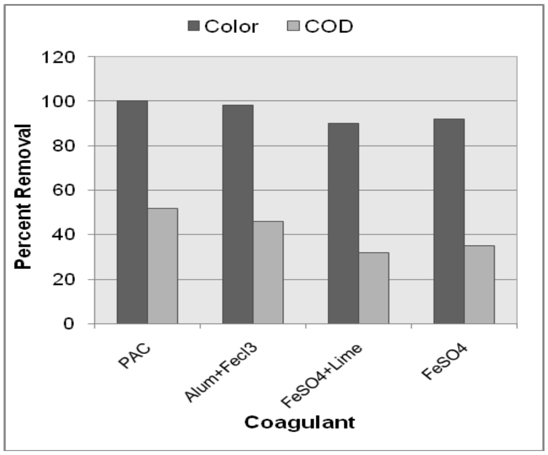

Figure 6 Optimum results of leachate treatment by chemical coagulation with various coagulants

Effect of Coagulants on removal of Metal and Heavy Metal Concentration

The removal efficiency of heavy metal by using different coagulants were analyzed and presented in Table 3.

Table 3 Characteristics of leachate concentration before and after treatment

Nitrogen, metal and heavy metal concentrations	Initial Value	Percentage removal of heavy metals by using different coagulants			
		PAC	Alum with $FeCl_3$	$FeSO_4$	$FeSO_4$ with Lime
Sodium (Na)	950	98	94	69	53
Potassium (K)	720	70	74	28	32
Calcium (Ca)	240	31	27	55	46
Magnesium (Mg)	130	62	76	52	50
Total kjeldahl nitrogen (TKN)	100	40	45	40	27
Ammonia nitrogen (NH_3-N)	47	51	46	39	35
Chromium (Cr)	19	74	90	64	56
Lead (Pb)	5	76	75	80	72
Zinc (Zn)	5.1	63	69	61	48
Nickel (Ni)	4	68	55	25	35
Copper (Cu)	37	46	49	44	37
Total Iron (Fe)	5	60	68	32	25

In case of heavy metal, PAC and Alum with $FeCl_3$ gave better removal efficiency with respect to $FeSO_4$. The removal efficiency was moderate by using $FeSO_4$.

CONCLUSIONS

Based on the factorial experiment results it ran an optimization study to identify the optimal operating conditions for coagulant dosage and pH. After analyzing all the results, it can be summarized that PAC exhibited good performance in removing color from leachate with lower dosages of coagulant. Moreover, results which obtained from Alum with $FeCl_3$ is also satisfactory. For 100% removal of color PAC needs 1g/L dosage at pH 6. Alum with $FeCl_3$ needs 1.4 g/L dosage for 98% removal of color at pH 6. 90% removal of color achieved by $FeSO_4$ with lime at pH 12 with a dosage of 2.7 g/L and 92% removal of color obtained by $FeSO_4$ at pH 12 with a dosage of 2.4 g/L. At optimum conditions, 52% COD removal was achieved using PAC, whereas removal using alum with $FeCl_3$ was 46%. So, higher removal efficiency in COD was found when the removal of color is higher. In contact, PAC and Alum with $FeCl_3$ gave better removal efficiency with respect to $FeSO_4$. So, it can be concluded that, PAC and Alum with $FeCl_3$ exhibited the best results than the other coagulants.

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Assessment of Bioavailability and Speciation of Zinc, Copper, Nickel and Chromium during Water Hyacinth Composting

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ABSTRACT

The bioavailability of heavy metals plays an important role in the toxicity of heavy metals during composting following land application. The potential toxicity risk from heavy metals depends on their chemical speciation. Therefore, studies were carried out on bioavailability and speciation of heavy metals (Zn, Cu, Ni and Cr) during 30 days agitated pile composting of water hyacinth (Eichhornia crassipes). The Tessier sequential extraction method was employed to investigate changes in heavy metals speciation during composting. Results showed that, the total metal concentration was increased during the composting. The water soluble heavy metals were reduced and water soluble Ni was not detected during the composting. Reducible and oxidizable fractions of Ni were not found during water hyacinth composting. From this study it can be concluded that the appropriate proportion of cattle manure addition significantly reduced the mobile and easily available fractions (exchangeable and carbonate fractions), and increase the residual fraction during the composting process. The residual fraction is more stable form and considered as unavailable for plant uptake.

Keywords; Composting, heavy metals, bioavailability, speciation

INTRODUCTION

Water hyacinth (*Eichhornia crassipes*) is a free floating macrophyte and most commonly used plant in constructed wetlands because of its fast growth rate and large uptake of nutrients and contaminants (Rai, 2009). Composting followed by land application represents one of the most economical ways for the treatment and final disposal of water hyacinth because it combines material recycling and biomass disposal at the same time (Villasenor et al., 2011). However, the presence of non-biodegradable and high level of toxic heavy metals in the compost frequently hinders agricultural land application. Uptake of heavy metals by plants and subsequent accumulation along the food chain is a potential threat to animal and human health (Wong and Selvam, 2006).

The bioavailability of metals in soil is a self-motivated process that depends on explicit combinations of chemical, biological and environmental parameters. These include soil properties such as pH, organic matter (OM) content, redox potential, cation exchange capacity, sulphate, carbonate, hydroxide, soil texture and clay content (Prabpai et al., 2009; Guala et al*.,* 2010). The pH, OM content and bioavailability of heavy metals are the major critical factors for heavy metal accumulation by both plants and animals (Li *et al.,* 2010).

The water soluble fractions are positively more biologically dynamic and it has the highest prospective of contaminating food chain, surface water and ground water (Iwegbue *et al.,* 2007). Metals in the water-soluble fraction may be readily leachable and bioavailable in the environment (Liu *et al.,* 2008). The total metal concentration obtained after strong acid digestion of compost sample is useful as an overall pollution indicator, but it does not provide useful information about the risk of bioavailability, which depends on their chemical form (Walter et al., 2006). Chemical speciation or sequential extraction of heavy metals from compost is a useful technique for determining the chemical forms in which these are present (Walter et al., 2006).

The objective of this work were to evaluate the water solubility and plant availability of heavy metals (Zn, Cu, Ni and Cr) and to assess the chemical forms of heavy metals accordance with sequential extraction method in the course of 30 days water hyacinth composting.

MATERIALS AND METHODS

Feedstock materials

Water hyacinth, cattle (cow) manure and sawdust were used for the preparation of different waste mixtures. Water hyacinth was collected from the Amingoan industrial area near Indian Institute of Technology Guwahati campus. Cattle manure was obtained from dairy farm near the campus. Sawdust was purchased from nearby saw mill. Prior to composting, the maximum particle size in the mixed waste was restricted to 1 cm in order to provide better aeration and moisture control.

Agitated pile composting

Different waste combinations were formed into trapezoidal piles (length 2100 mm, base width 350 mm, top width 100 mm and height 250 mm, having length to base width (L/W) ratio of 6. Agitated piles contained approximately 150 kg of different waste combinations (90kg water hyacinth, 45kg cow dung and 15kg sawdust) and were manually turned on 3, 6, 9, 12, 15, 18, 21, 24, 27 and 30th day. Composting period of total 30 days was decided for agitated pile composting. Homogenized samples were collected from five different locations in the pile on 0, 6, 12, 18, 24 and 30th day.

Analysis of physical and chemical parameters of the sample

Temperature was monitored using a digital thermometer throughout the composting period. About 500 g of each grab samples were collected from five different points. Finally all the grab samples were mixed thoroughly to make a homogenized sample. Triplicate samples were collected were dried at 105°C in oven for 24 hours and moisture content was calculated, dried samples were ground to pass to 0.2 mm sieves and stored for further analysis. Each sample was analyzed for the following parameters: pH, conductivity (1:10 w/v waste: water extract), organic matter (Kalamdhad et al., 2009). Water-soluble heavy metals are determined after extraction of 2.5 g of sample with 50 mL of distilled water (sample:solution ratio = 1:20) at room temperature for 2 h in a shaker at 100 rpm (Ciavatta *et al.,* 1993). Total metals (Zn, Cu, Mn and Ni) were determined by Atomic Absorption Spectrometer (Varian Spectra 55B) after the digestion of 0.2g sample with 10 ml of H_2SO_4 and $HClO_4$ (5:1) mixture in block digestion system (PELICAN EQUIPMENTS Chennai-India) for 2h at 300°C.

Methodology of sequential extraction

The conventional method was designed and developed by Tessier (Venkateswaran et al., 2007) for heavy metal speciation into five species. The extraction was carried out with an initial mass of 1.0 g oven dried sample in polypropylene centrifuge tubes of 50 mL capacity. After each successive extraction, the supernatant liquid was removed with a pipette after centrifugation at 10,000 rpm for 5 min and made up to required volume for analysis of heavy metals. The residue was washed with 20 ml of Milli Q water by shaking for 15 minutes followed by centrifugation without loss of solids. The extracts were stored in polythene bottles for metal content determination. All extractions were performed in triplicate and the mean value was presented with standard deviation. The following steps were adopted: (1) Exchangeable (F1): About 1 g sample was extracted at room temperature with 8 mL of 1 M $MgCl_2$ (pH 7) with continuous agitation for 1 h. (2) Carbonate (F2): Residue from above step (1) was leached at room temperature with 8 mL of 1 M of NaOAc (pH 5 adjusted with conc. HOAc) with continuous agitation for 5 h. (3) Reducible (F3): Residue from (2) was extracted with 20 mL of 0.04 M $NH_2OH.HCl$ in 25% (v/v) HOAc agitated for 6 h at 96ºC. (4) Oxidizable (F4): To the residue from (3) was added with 3 mL of 0.02 M HNO_3 and 5 mL of 30% H_2O_2 (pH 2, adjusted with conc. HNO_3); heated at 85 ºC for 2 h with occasional agitation. A second 3 ml aliquot of 30% H_2O_2 was added and heated at 85 ºC for 3 h with occasional agitation. After cooling, 5 mL of 3.2 M NH_4OAc in 20% (v/v) HNO_3 was added; diluted to 20 mL; agitated for 30 min and centrifuged. (5) Residual (F5)**:** Residue from (4) was digested with 10 ml of H_2SO_4 and $HClO_4$ (5:1) mixture in block digestion system (PELICAN EQUIPMENTS Chennai-India) for 2h at 300°C.

RESULTS AND DISCUSSION

Physicochemical analysis

The composting pile temperature went through three typical phases (heating, thermophilic and cooling phase) and ranged from 26°C to 56°C during the entire period of composting. However, the cattle manure affected the temperature during different composting phases. Moisture loss during the composting process can be viewed as an indicator of decomposition rate, because the composting material requires optimum moisture content in it for the organisms to survive (Kalamdhad et al., 2009). Thus the organic matter content of the composting mass decreases as composting proceeds (Singh et al., 2009). Table 1 shows the moisture loss, organic matter loss, conductivity reduction and enhancement of pH during the process.

Table 1 Physicochemical parameters during the composting process

Days	Moisture content (%)	Conductivity(dS/m)	pH	Organic matter (%)
0	83.9±0.06	6.4±0.04	6.4±0.03	78.3±0.05
6	82.8±0.02	6.3±0.00	7.3±0.01	74.7±0.01
12	72.7±0.04	6.1±0.03	7.2±0.01	73.3±0.00
18	59.6±0.57	5.7±0.04	7.3±0.00	69.6±0.64
24	54.4±0.04	5.8±0.06	5.5±0.30	65.4±0.05
30	37.6±0.06	4.7±0.04	7.3±0.01	60.3±0.06

Heavy metal speciation during composting

Table 2 illustrates the total concentration of metals (Zn, Cu, Ni and Cr) during the composting process. These heavy metals were concentrated during the composting process due to weight loss in the course of composting. Water soluble fractions of organics and metals are the most readily bioavailable in compost applied to soils (Hsu and Lo 2001). The water soluble concentrations of Zn was reduced from, 2.1% to 1.5% of total Zn at the end of the composting period (Table 2). The water soluble concentration of Cu was reduced from 3.6% to 2.2% of total Cu at the end of the composting period (Table 2). The groups of –OH and –COOH supplied by cattle manure increased the binding sites and combined with Cu to form insoluble and immobile complexes, thus the concentration of free Cu^{2+} decreased and the potential environmental risk was drastically reduced (Guan *et al.,* 2011). The water soluble concentration of Cr was reduced from 0.8% to 0.2% of total Cr at the end of the composting period (Table 2). The oxidation process and the formation of organo-metallic complexes taking place during composting could reduce the soluble contents of metals (Fang and Wong, 1999). The water soluble Ni was not detected throughout composting process.

Table 2 Total heavy metals concentration during the composting process

Days	Heavy metals concentration			
	Zn	Cu	Ni	Cr
0	161.1±1.6	31±0.5	187.3±1.8	257±5.0
6	200.8±0.2	45.5±0.5	222.8±3.3	259.3±19
12	282.8±7.2	51.8±0.8	293±2.0	261±4.5
18	272.5±2.0	46.8±0.8	319.5±5.5	236±3.8
24	235.9±1.0	49±0.5	284.5±9.0	270.5±5.5
30	297.8±3.0	103.3±0.8	235.8±1.8	279±1.3

All the movable fractions of Zn (from F1 to F4) were reduced (percentage of total fraction) in comparison to other trials, however F5 fraction was rapidly decreased in initial phase of composting but increased at the end of composting (Fig. 1). The reduction of all movable form may be due to the formation of Zn complex with humic substances formed at the maturity stage of compost. Humic substances contain various organic functional groups that can sorb metal ions through ionic force (Cai et al., 2007).

Table 3 Water soluble heavy metals concentration during the composting process

Days	Heavy metals concentration			
	Zn	Cu	Ni	Cr
0	3.4±0.04	1.1±0.04	ND	2±0.07
6	2.0±0.01	1.82±0.02	ND	1.7±0.07
12	8±0.06	1.8±0.04	ND	1.8±0.02
18	7.42±0.22	2.1±0.06	ND	1±0.11
24	8.4±0.06	1.84±0.08	ND	1.4±0.06
30	4.5±0.13	2.25±0.01	ND	0.54±0.04

ND- Not detected

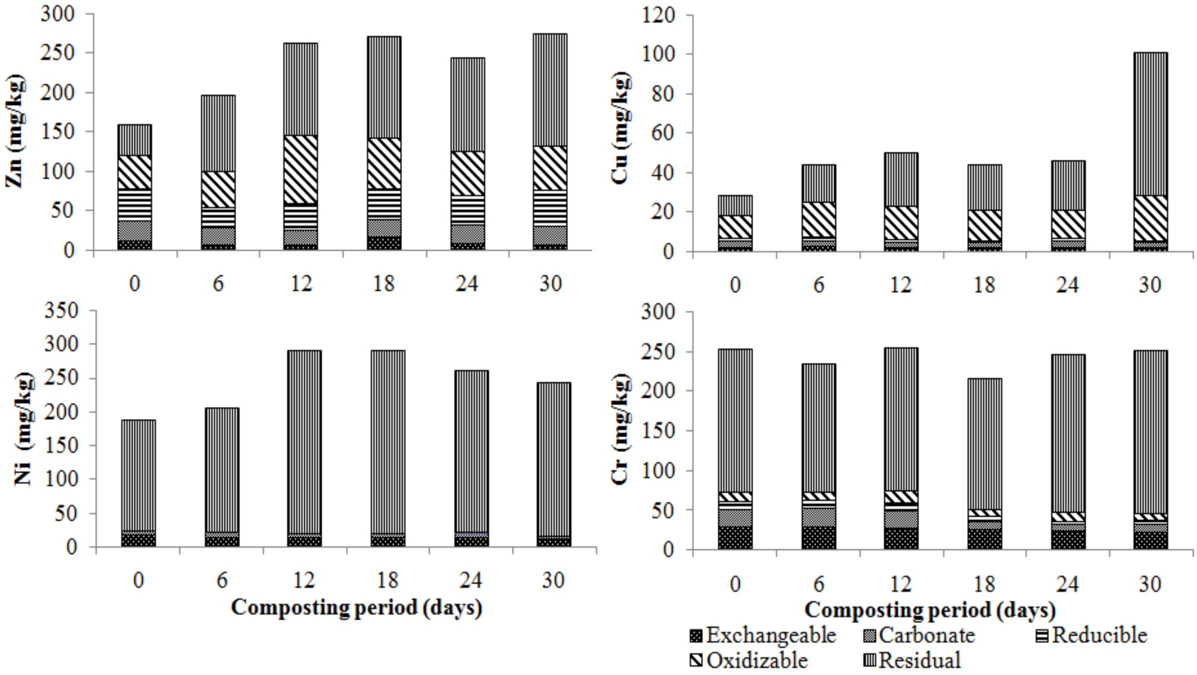

Figure 1 Speciation of heavy metals during water hyacinth composting

The F1, F2, F3 and F4 fractions of Cu were decreased, however the F5 fraction was increased of the total fraction of Cu during the composting process (Fig. 1). A maximum reduction of F1 and F2 fractions of Cu were observed from 7 and 10% to 1.6 and 2.5% respectively which may be due to the formation of Cu ion complex with two or more organic functional groups mainly carboxylic, carbonyl and phenolic, so that the ion is immobilized in a rigid inner-sphere complex (Qiao and Ho 1997). The F1 and F2 fractions of Ni were decreased, but the F5 fraction was increased of the total fraction of Ni during the composting process (Fig.1). The F3 and F4 fractions of Ni were not detected in all trials during the composting period. By composting process, the proportion of F1 and F2 fractions of Ni were decreased while the proportion of F5 fraction of Ni was increased; it was due to F1 and F2 fractions of Ni were transformed to the F5 fraction during the composting process (Zheng et al., 2007). The F5 fraction of Cr was increased whereas F1, F2, F3 and F4 fractions were reduced from 11.5, 9.1, 3.6 and 4.6% to 9.1, 3.7, 2.4 and 2.8% of total fraction of Cr during the composting process (Fig. 1). The significant reduction in fractions (F1- F4) was observed during the composting process might be explained that, the F1 and F2 forms may bound with various organic functional groups present in humic substances, while the F3 and F4 fractions might be converted into F5 fraction during the stabilization of compost.

CONCLUSION

The water solubility of Zn, Cu and Cr was reduced but Ni was not detected during the composting process. The concentration of Cu was very low compared to the other metals, but the percentage of

F1 and F2 fractions were similar as other metals. The F3 and F4 fractions of Ni were not detected during the water hyacinth compost process. The maximum reduction in F1 and F2 fractions of all metals were found during the process, which are more toxic and easily bioavailable fractions. The cattle manure addition significantly reduced easily available fractions (F1 and F2) due to better humification.

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Development of a Dry Digestion Biogas Plant in Kumasi, Ghana

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ABSTRACT

Dry anaerobic digestion is considered a promising technology for treatment of organic waste in developing countries, mostly due to its simple design which can be constructed and operated at low cost. As real-scale experience is missing, a dry anaerobic pilot digester was developed by Eawag in collaboration with KNUST and ZHAW by converting a second-hand shipping container into a biogas reactor. The digester was filled with organic solid waste from the dumpsite and four batch runs of 4-8 weeks were conducted in Kumasi, Ghana. First batch results revealed insufficient gas tightness of the container and unsatisfactory biogas production. After several modifications and tests, a second biogas digester was built using a shipping container of higher quality. Finally, the fifth batch in this new digester showed an increasing gas production and methane content of 55-60%. However, gas production levels were still much lower than expected.

INTRODUCTION

Background

Similar to many other developing countries also in Ghana the waste and energy sector are confronted with major challenges. In 2009 nearly 60% of Ghana's primary energy demand was provided by firewood and charcoal, which is associated with deforestation and indoor air pollution (Yankey, 2009). Moreover, only 15% of the waste in Ghana is treated in an adequate manner (KMA, 2000). Indiscriminate waste disposal in the streets or in open dumps leads to severe environmental pollution and risks for human health. As municipal solid waste in Ghana consist of up to 60% of biodegradable organics (NESSAP, 2010), there is an enormous potential for the production of biogas that could be used for cooking, lighting or electricity generation. Furthermore, the digestate can be composted to obtain a nutrient rich fertilizer for agriculture.

Dry anaerobic digestion – discontinuous process with percolation

The formation of biogas through dry digestion occurs according to the principle of anaerobic digestion. The difference between wet and dry digestion is in terms of the water content of the feedstock in the reactor. This is specified by the percentage of total solids (TS) of the total mass in the reactor. Previous research characterizes dry digestion by a percentage of TS, between 22-40% (Ward et al., 2008) or above a TS of 15% (Li, Park & Zhu, 2011). However the main criterion is the feedstock stackability. In a dry digestion process the organic solid waste is filled batch-wise into gastight garage-like digester units. The fresh organic material is inoculated with old material from a previous digestion batch and/or cow dung. Once filled, the digester remains closed throughout the digestion phase which lasts for 4-8 weeks, depending on the characteristics of the substrate. During this time the digester content is not turned. A percolation system allows collection of leachate from the feedstock and continuous or periodical sprinkling of this same leachate back onto the feedstock. Percolate sprinkling is stopped a few days before opening the digester to ensure the material is well drained for subsequent further use. Before opening the digester, the digester is flushed with CO_2 to avoid an explosive methane-air mixture after opening and during emptying of the digester. The treatment plant can easily be extended by one or more digester modules. This allows better control and optimization of retention time to ensure improved gas yield and flexible adjustment to changing input materials.

Problem statement

Dry digestion approaches have already been implemented successfully for a number of years in Europe (Lutz, 2010; Bioferm, 2011). Experts report a high potential for application in developing countries as it has a very simple design which can be constructed and operated at low cost, does not need substantial addition of water, requires low process energy consumption and because the residues after digestion can be more easily treated in a safe way (Fei-Baffoe, 2006; Khalid et al., 2011). Nonetheless, to date, there have been no documented experiences available with this technology used in a developing country context. As the commercially available systems implemented in Europe are expensive and complex in their operation they would most probably fail if transferred directly to a developing country situation, it was thus considered important to gain practical experience with the construction and operation of an adapted, locally developed dry digestion system and thereby prove its appropriateness for low and middle-income countries.

This paper presents the results of the research on dry digestion conducted in Kumasi, Ghana between December 2010 and April 2012. The objective of this research was to explore the possibilities for construction of a dry digestion pilot plant in developing countries using local materials and to conduct several test-runs with the constructed pilot plant under local everyday conditions of Kumasi, Ghana. The development, building, test-operation and lessons learnt with the pilot dry digestion plant are hereby presented.

METHODOLOGY

- The 1st phase (Dec. 2010 - May 2011) of research was conducted by Burri and Martius (2011). Like all following phases it took place in close collaboration with researchers of Kwame Nkrumah University of Science and Technology (KNUST) in Kumasi, Ghana and technical support of Zurich University of Applied Sciences (ZHAW). Research started with a literature review, a visit to a dry fermentation plant in Switzerland and consultation of experts. In a preliminary study the core technical elements of a dry digestion plant were identified. In small-scale experiments with a 300L high-density polyethylene barrel, technical solutions regarding percolation system, gas tapping system and air tightness were studied on-site in Kumasi. The suitability of different materials and designs for developing countries were evaluated. Based on this assessment a dry digestion prototype plant was conceptualized, constructed and tested (batch #0) with fibrous feedstock originating from the dumpsite in Kumasi.
- The 2nd part of the research study (Feb.– Nov. 2011) included different experiments regarding hydraulic characterization, percolate flow experiments, inoculate breeding and air tightness. These tests and the subsequent batches (#1 and #2) were documented by Biolley and Diggelmann (2011).
- In order to optimize the digestion process, the 3rd phase (Nov. 2011 – May 2012) comprised improvement of pilot plant before experimenting with batch #3. Based on the experiences and results the decision was then taken to construct a new pilot digester and perform a further batch (#4) experiment (Robbiani, 2012).

During the batch experiments, the following parameters were measured:

- Total solids (TS,) and Volatile Solids (VS, both according to Clesceri et al., 1998) of the feedstock and the digestate
- Temperature, pH, Redox potential, withal measured using a Hach HQ D40 pH measuring sensor, Electrical Conductivity (Cyber Scan PC300) and Chemical Oxygen Demand (COD, open reflux titration) of the percolate
- Gas production (Gas meter type G4ZR W C-0 with minimal gas flow of 0.04 m3/h) and gas composition (measured using a Dräger X-am 6000)

RESULTS

Construction

Four core technical elements that are essential for the functionality of a dry digestion system are:

- Air tight shell: The airtight digester is the core piece of a dry digestion plant as it provides the anaerobic environment needed for the biological process of biogas formation. Moreover it has to prevent leakage of the produced gas. In contrast to plants operating in Europe dry digestion systems in tropical climate do not require installation of a heating system.

- Opening: The door has to fulfil two main tasks. One is to enable easy loading of the feedstock and unloading of the digestate. The other one is, when closed, to seal the container airtight and avoid frequent and costly repairs after each opening.
- Percolation system: In addition to anaerobic conditions methane forming bacteria require a humid environment (with dry digestion a water content of approximately 70%) and this should be evenly distributed in all parts of the substrate. This can be realized with a percolation system that circulates the effluent from the feedstock (leachate) which contains a population of anaerobic bacteria (percolate). A shower-like installation at the roof of the digester sprinkles the percolate over the biomass. The percolate trickles down, exits at the bottom and can be collected again. Before the percolate is re-circulated, it passes through an appropriate filter to prevent coarse particles from entering and blocking the circulation system.
- Gas tapping system: The gas tapping takes place at the highest part of the digester from where the produced biogas is chanelled through a pipe to the outside of the digester. Commonly the gas tapping system is equipped with a flash back arrestor and a dewatering unit. The flashback arrestor stops sparks and flames from entering the container and igniting a possibly explosive gas mixture. The dewatering unit extracts condensation water from the outflowing gas. The tapped gas then passes through a gas flow meter and gas composition meter before being directed to its end use (in this research the biogas was simply released).

Figure 1 presents the scheme of a dry digestion plant with its four core components described above.

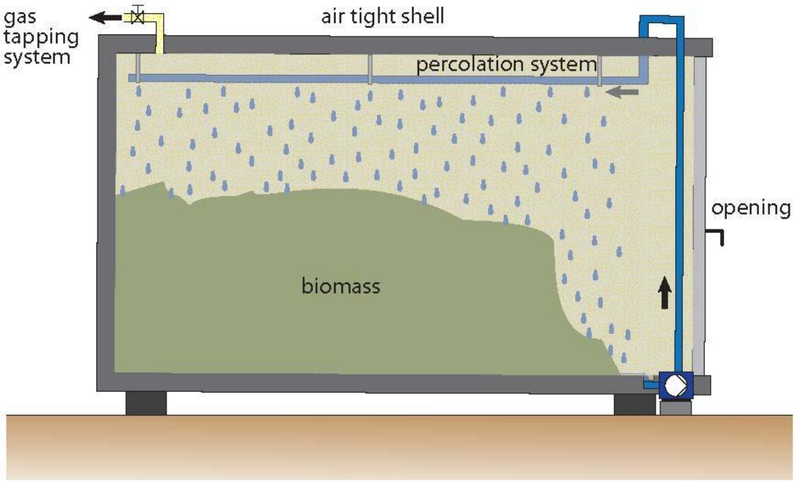

Figure 1: Scheme of a dry anaerobic digestion system

The suitability of materials and design options were evaluated based on the criteria listed in Table 1.

Table 1: Criteria to assess suitable materials for building of digester

Material aspect	**Criteria**
Economics	• Low material cost • High income generation for local enterprises and individuals
Availability	• Available within Ghana, preferably Kumasi • Little time needed to obtain items
Durability	• Not subject to corrosion • Withstand weather and other external influences
Security	• Suitable for inflammable gas
Ease of handling	• Low degree of needed expert knowledge or specialized tools needed
Form	• Suitable for easy loading and unloading • Airtight opening can be built easily

For building the pilot dry digester three potential materials were taken into account for further investigation: Polyfoil, a concrete structure or shipping container. Table 2 reveals their advantages and disadvantages.

Table 2: Positive and negative aspects of different design options

Option	**Positive aspects**	**Negative aspects**
Polyfoil	• Cheap • Easy and quick to build the digester • Easy to remove after project's end • Lightweight makes it easy to transport • Digester room can serve as gas storage if dimensioned properly • Easily scalable to required size • Resistant against corrosion	• Low resistance against mechanical damage • Loading and unloading difficulties • Technical equipment and knowledge needed for plastic welding & gluing • Possible electrical discharges and sparks from polyfoil can lead to ignition of burnable gas mixtures
Concrete structure	• Scalable to the desired size • Different shapes possible • Different qualities of concrete and bricks available • Construction knowledge available • Durable material	• Not mobile • Air tightness problems and pressure cracks can arise from bad quality work or material • Airtight door to be custom made • Corrosion damages can occur • Building of digester needs time and adds uncertainties
Shipping container	• No need to build the digester • High resistance against mech. damage • Big opening for loading and un-loading • Standardized, worldwide use • Equipped for easy transportation	• Standard two wing door can lead to air leakages • Big quality differences amongst second hand containers • Corrosion if not maintained properly

The criteria from Table 1 were used to assess the suitability of these three options (Table 3)

Table 3: Ordinal score (one to five) of each option with respect to the criteria listed in Table 1.

	Polyfoil	**Concrete**	**Shipping container**
Economics	4	3	3
Availability	4	3	4
Durability	1	3	3
Security	3	4	4
Ease of Handling	3	3	5
Form	4	4	4
Sum	**19**	**20**	**23**

The final choice was to build the prototype digester with a 20 foot (6.1 m) second-hand shipping container (Fig.2)

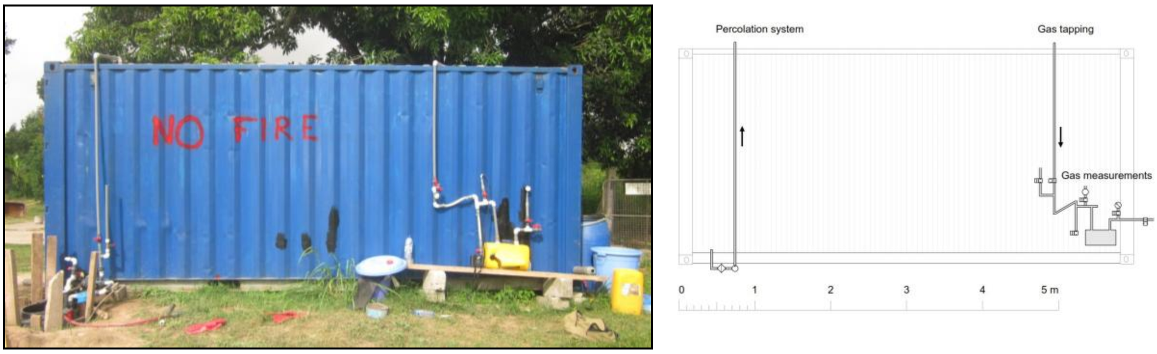

Figure 2: Side view of first prototype dry digester with schematic representation (Burri & Martius,2011)

Regarding the criteria of Table 1 the chosen shipping container performs as follows:

- Economics: The price for second hand shipping containers can vary considerably (800 - 8000 US$ depending on the quality). The container used for the first prototype cost 1200 US$.
- Availability: Ghana has major harbor cities (Tema and Takoradi) and thus second-hand shipping containers are abundant. These only have a limited lifetime in which they get a certificate for oversea shipping. After that they are still in good shape for being reused. Transport within Ghana can easily be arranged through shipping companies or private truck owners.
- Durability: Shipping containers are made from steel and are painted with long-lasting bitumen paint to withstand rough conditions. However, cracks in the paint are susceptible to corrosion. Thus it needs to be well maintained and repainted from time to time. The floor is made of wood panels that are joined together with silicon sealing to grant air tightness. As wood panels and sealing degrade over time they have to be replaced on a regular basis.
- Security: There is no risk of sparks due to electrical discharges as when handling polybags. However, sparks can occur while opening and closing doors due to friction between metal parts. This needs to be particularly avoided when there is a risk of an explosive mixture in the digester (6-12% methane in air, Deublein and Steinhauser, 2008). If the outlets are blocked, high overpressure could result in ruptures or cracks.
- Ease of Handling: A shipping container can be modified with ordinary metal working tools. Welding machines, electrical grinders and skilled workers are available in every bigger city in Ghana. Substrate loading and unloading can easily be carried out through the big steel doors. A shipping container biogas plant can be built at a location with access to skilled workers and machines and then be transported to the operation site.
- Suitable Form: Shipping containers are mainly available in two sizes (20’ and 40’ (6.1 m and 12.2 m). As both sizes have a big door, loading and unloading can easily be done. One drawback is the two wing door, as the connection point of those two door wings in the middle causes air tightness problems.

After a first trial with organic waste from the landfill, it became obvious that improvements were necessary regarding the digester’s air tightness. Main reason for the lacking air tightness proved to be the two wing door of the container. Different airtightness tests (compressor/outgoing gas flow test, manometer for detection of max. digester pressure, smoke test) and experiments were conducted trying to identify leakages and improve air tightness. A garden hose was installed to tighten the two wing door. Furthermore a wooden frame was constructed behind the doors to improve the sealing and combined with bicycle tire which were glued on to the wood where necessary. The bicycle tires were then pumped with air to increase sealing, but the results were not satisfactory. Thus, as a solution to assist the leaking tube, silicon sealant was applied when closing the door for the first test run.

Further problems during the first trial included malfunctioning of the percolate pump due to foam building. As a consequence the percolate storage tank was removed and a direct connection from the outlet to the filter was constructed. The sprinkling holes were enlarged to prevent clogging by solids that pass the installed filter. The permanent use of electricity for the percolation pump made the installation of a separate electricity connection inevitable.

All the piping and connections were made with 1’’ (2.5 cm) PVC plumbing material, which is widely available in Ghana. The percolation system and gas tapping systems of the first prototype are shown in Figure 3.

Figure 3: Percolate sprinkling, gas measurement bench, scheme of gas measurement (left to right)

Batch #0 (test run)

The digester was loaded with 8 t organic fraction of municipal solid waste, inoculated by a mix of fresh cow dung (1.05t), sheep dung (0.45 t), fresh sewage sludge (0.2 t) and inoculated percolate (1100 L water and 750L liquid cow dung). The percolate mix provided an appropriate pH of 6.5. Pumping and sprinkling of percolate onto the feedstock was continuously controlled by a time switch. During this test run, within one day the effluent exiting the reactor dropped to a pH of 5.6 and remained almost constant even though sodium hydroxide (NaOH) was added before it was recirculated. This drop in pH and the constant acidic conditions indicate an unfavorable environment for methane-forming bacteria. In fact the methane content in the produced biogas was very low and stabilized around 21% at the end of the test run after 23 days. The low level of overall generated gas volume measured can be explained by either a very low biological process rate or leakage and therefore an unaccounted gas production. The total of 15 m^3 biogas over a period of 23 days is very low. A regular plant can be expected to produce around 9 m^3 per day (Gronauer and Aschmann, 2003).

Batch #1

The feedstock used for the next batch was 5 t of organic solid waste, inoculated by 4 t of fresh cow dung from the abattoir and an inoculated percolate (as in Batch #0). During this test run, the pH again remained below 7.5 and was predominately around 5.5, even though calcium carbonate ($CaCO_3$) was added to the percolate to increase its pH. After 32 days the biogas production was 4.5 m^3 and methane content in the biogas reached 18%, but with a sharply rising trend. After 32 days however the container was unloaded, cleaned and maintained to prepare for the next batch. The digestate was dumped on a pile near the digester and half of it was used as inoculum for the second batch. The percolate was stored in buckets so, after pH adjustment, it could be used for the next fermentation process.

Batch #2

For the second batch, the main features of the setup and monitoring was not changed, but the feedstock was inoculated differently. The digestate of batch #1 was used as inoculum, and the ratio of the compounds was approximately 3:1:2 (old digestate: feedstock: cow dung) based on wet weight. Cow dung and organic waste were considered as feedstock for this second batch and represent about 50% of the total amount. This ratio is generally recommended as inoculation for dry fermentation (Gronauer and Aschmann, 2003). The pH value of the percolate dropped from 7.1 to 5.7 in the first week but then stabilized around at pH of 7.7 after 45 days. During the acidic initial phase a total of 6 kg NaOH (4 x 1.5 kg within seven days) and 4.5 kg $CaCO_3$ was added to the percolate to increase its pH. The temperature of the percolate was in the range of ambient temperature between 25 and 34°C. The methane quantity reached the max. of 50-60% after 25 days and then stabilized at around 40%.

Before starting with the third test run, some adjustments to the setup were necessary. After unloading the digestate from batch #2, the digester was cleaned and the percolation system tested with plain water. Silicone sealant was applied on the container floor to seal the cracks through which the liquid and gas could possibly leak. Moreover, in order to prevent corrosion, black anti-corrosive paint was applied on the outer and inner walls of the container.

Batch #3

For the third batch experiment, 5 t of fresh organic waste was mixed together with 6 t of digestate from the previous batch. In addition, 1.8 t of cow dung was added as extra-inoculum and 5 kg of $CaCO_3$ was added to 1500 L of percolate to stabilize the pH (initial pH: 8.15). During the test run the pH remained between 8 and 8.5, which is in the optimum range for anaerobic digestion and the methane content stabilized between 55 and 60%. However, very little gas production was observed. The installed gas meter had been subject to corrosion and was notable to measure the low gas pressure and flows properly. Sufficient gas pressure could not be attained probably because of leakages around the opening.

Figure 4 shows both TS and VS of the substrate before and after 45 days of digestion . Although a reduction can be observed, the value for VS is still high after digestion, which suggests that a relatively high quantity of biodegradable solids are not yet degraded and remain present in the digestate. This can be explained by a high amount of hardly degradable lignin in the feedstock or that some materials of the feedstock require a longer digestion time.

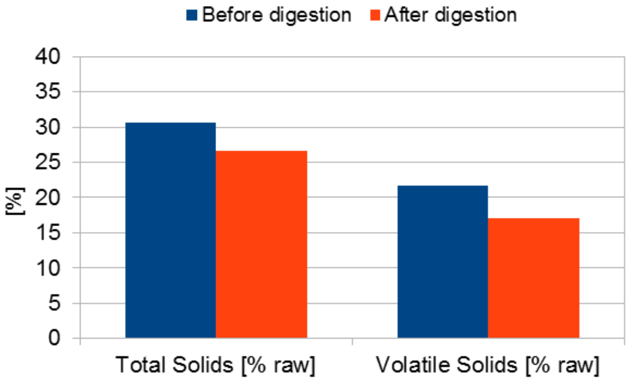

Figure 4: TS and VS of feedstock before and after digestion period (45 days)

An additional problem was the loss of percolate from the digester. To make up for these losses around 300 L were added in total during the digestion period.

Installation of a new digester

Based on the results and problems faced during the test batch runs #1 to #3 the decision was taken to build a new digester to improve air tightness. Again, a 20' shipping container was chosen but of much higher quality. The investment costs for the new container amounted to 8000 US$. The investment costs for all additional equipment (pump, pipes, valves, filters) was approximately 200 US$. The design was based on the first digester and all equipment was again locally available material and constructed on-site. The digester is equipped with metallic inlet and outlet pipes welded directly to the surface of the container and connected to PVC pipes for the percolate circulation and for the gas outlet. To protect the container from rainwater and corrosion a roof was built. In addition, the digester was mounted on supports. The elevated position prevents the wooden plates at the bottom of the container from rotting when in contact with the soil. Moreover, in the elevated position the operators were able to easily locate leakage of percolate and subsequent collection of this leaking percolate. Cement blocks were placed at one end of the digester to create an adequate slope allowing the percolate to accumulate downslope of the digester to then be more easily collected and pumped for recirculation. To prevent percolate leaking from the bottom of the digester, fiberglass coating was applied to the floor, thereby sealing the interstices between the wooden plates. Finally, anticorrosive paint was applied on the inner and outer walls.

Figure 5: Second dry digester made of higher quality shipping container

Batch #4 (new digester)

For batch #4 6 t of fresh organic waste was mixed with 6.4 t of digestate from the previous batch. After 30 days of digestion methane content reached 55-60% and the pH stayed between 7.4 and 8.5, indicating ideal environment for methane-forming bacteria. Nevertheless, the measured amount of biogas (6.6 m^3 in 30 days) was very low. Based on the achieved VS reduction of 609 kg during the digestion period, the calculated biogas yield for batch #4 (0.011 m^3 / kg VS) represents only 5.5% of the biogas yield that literature reports for similar substrates (0.2 m^3 / kg VS, Eder and Schulz, 2006).

An overview of the parameters measured during all batch test runs is shown in Table 4.

Table 4: Overview of test-run results

	Batch #0	**Batch #1**	**Batch #2**	**Batch #3**	**Batch #4**
Feedstock (org. solid waste)	8 t	5 t (10 m^3)	2.4 t (6 m^3)	5 t (10 m^3)	6 t (12 m^3)
Inoculum					
- cow dung	1.05 t	4 t (4.44 m^3)	4.05 (4.5 m^3)	1.8 t (2 m^3)	-
- sheep dung	0.45 t		-	-	-
- sewage sludge	0.2 t	-	-	-	-
- digestate of previous batch	-	-	6.3 t (7 m^3)	6 t (7.5 m^3)	6.4 t (8 m^3)
Percolation liquid (total)	1850 L	1000 L	1500 L	1500 L	1500 L
- fresh water	1100 L				
- cow dung liquid	750 L				
Retention time	23 days	32 days	45 days	45 days	30 days
pH (percolate)	5.6 – 7.5	6.5 – 7.5	5.8 – 7.3	6.5 – 8.5	7.5 – 8.5
pH buffer					
- $CaCO_3$	-	3 kg	4.5 kg	5 kg	4 kg
- NaOH	n.a	-	6 kg	-	-
Temperature	25-38°C		25 – 34°C		27 – 37°C
Biogas production					
- total	15 m^3	4.5 m^3	n.a	n.a	6.64 m^3
- per day	0.65 m^3	0.14 m^3	-	-	0.22 m^3
CH_4 content (stabilized)	21%	18% (n.s.)	40%	55-60%	55-60%

(n.a.: not available/ measured; n.s.: not stabilized)

DISCUSSION

The low amount of measured biogas production can be explained in various ways.

- The *nature of the feedstock* (fibrous, cellulose and lignin rich material) is not so suitable for biogas production. This kind of feedstock has an ideal structure that allows good percolation of liquid through the mass without clogging outlet pipes, filters or the pump. However, lignin and cellulose rich materials are not easily degraded by anaerobic bacteria and thus not ideal for biogas production.
- Secondly, the he *substrate size* influences the rate of anaerobic degradation. The smaller the substrate size, the more easily it can be reached and degraded by the microorganisms. The feedstock is not shredded thus this reduces its potential to produce biogas, or reduces the rate at which biogas is produced. Shredding the substrate in small pieces would increase the rate at which biogas is produced but at the same time is an additional processing step which requires equipment and energy and therefore cost. Shredded fine material may also cause difficulties with the percolate circulation system (clogging of pump).
- Thirdly, the liquid percolate predominately contains the microorganisms that degrade the substrate. If the *percolate is not uniformly distributed* over the whole pile of feedstock some parts of the feedstock may not be well inoculated. After the batch test run #3, when checking the percolate circulation piping system, it was observed that some holes of the sprinkling pipes were clogged. This means that the feedstock was sprinkled at some determined places only. This may lead to preferential flowpaths through the feedstock . In locations where the feedstock is not in regular contact with the percolate biological activity is limited.
- Finally fourthly, the study cannot discard that *biogas may still be leaking* out of the digester. Although the digester was improved compared to the digester used in batch #3, the airtightness of the digester still remains an issue. Furthermore the *gas meter was not appropriate* due to its limitations with detecting low gas flows. There is therefore a chance that not all biogas produced was actually measured by the gas meter. The instrument measuring gas flow has a detection limit of 40 liters per hour. Any gas flow lower than that would not be detected by the gas meter.

CONCLUSION

Batch dry fermentation systems have a simple design and consume low amounts of water and process energy. They can use bulky organic solid waste with high dry matter content as substrate and the digestate can be easily composted given its lower water content compared to wet digestion systems. The construction of the pilot plant on the basis of a shipping container proved possible in Ghana with local material at a fairly low price and without need for complicated technical parts or specialized tools. However, the pilot plant revealed some drawbacks of the technology: The airtight sealing remains a major challenge. Particularly the sealing of the door needs further investigation to find a suitable permanent solution. The security risk of potentially explosive air/methane mixture is an important factor to consider and sparks have to be avoided by all means. Operation of the dry digestion facility is considerably safe if precautions are followed. However, construction, operation and maintenance needs well trained personnel to avoid faulty handling and prevent severe accidents. As the technology is yet not very mature, the security issue needs further research. This includes the development of a digester air space flushing system. Further research regarding durability of parts like the PVC piping or the sealed wooden floor is also required.

When comparing dry digestion to wet digestion technologies (such as fixed dome or floating-drum digesters), few arguments remain in favor of dry digestion. Wet digesters can also have a simple design without mixing or stirring device, hence no need for electricity. Their performance and suitability has been proven in a vast number of plants throughout Asia and Africa. Rough cost assessments show that the prices of wet and dry digester are similar. An advantage of dry digestion compared to wet digestion is the stackable substrate that can be used with high dry matter content which alleviates the problems of digestate post-treatment and use. The downside of this is however is a lower biogas yield. Although the performance of dry fermentation plants in Europe show a competitive edge, the added value in developing countries has yet to be proven.

At this stage of research, reasons for the low gas production still need to confirmed. It is necessary to better understand the ongoing dynamics inside the digester and determine the gas potential and suitability of the feedstock. Furthermore, methods to examine gas tightness of the container still need to be further developed. as well as a reliable gas measurement device which can also measure low gas flows.

With only one container and a retention time of 1-2 months, it was not possible to test and modify various parameters within a useful timeframe. The authors therefore recommended research with a larger number of smaller bench-scale plants to test different substrates, feedstock ratios, varying retention times, and different sprinkling practices of percolate. Another important aspects is to explore the market for the gas as well as the treated digestate and compare different digestate post-treatment methods to determine how demand for gas and digestate use influences the viability of such treatment facilities.

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Cost Effective Treatment for Textile Wastewater

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ABSTRACT

Textile wastewater treatment (WWT) a rigorous procedure and generally requires chemical procedure which makes it expensive.Industrial by-products such as fly ash from coal based power generation has been used in many part of the world considering their cost effectiveness and environmental benefits. Recently, coal based power plant at Barapukuria in Bangladesh showed a promising source for fly ash supply. Apart from landfill, fly ash has been used for various other purposes such as an addition in concrete, sub-base course for highway around the world. In this study textile wastewater sample was collected from Chittagong region to carry out laboratory experiments for identifying the environmental hazardous components. Experimental result gave unacceptable level of BOD and colour in these samples. A series of experiments were carried out with three different fly ashes and wastewaters to remove/reduce these components. The influence of (i) fly ash quantity (i.e. presence of activated carbon) and (ii) contact time were noted. In this connection, specific surface area rather than loss-on-ignition has observed as an indication for activated carbon in fly ash. This experimental study showed reasonable outcome on WWT and it is envisaged, with further study the approach could lead to large scale WWT.

Keywords: Wastewater Treatment (WWT); Fly Ash; Cost effective

Introduction

Textile wastewater is a serious environmental concern since they contains dye/colour. To avoid associated high expenses for treatment, nearby open water body such as flowing river became the common practice for the untreated wastewater disposing in many developing countries. The abnormal coloration of the recipient water body could have severe environmental consequences due to presence of a large number of contaminants such as toxic organic residues, acids or bases and inorganic contaminants. Thus cause rise of organisms, reduction of light penetration and photosynthesis (Malamis et al., 2011, Rastogi et al., 2008). In Bangladesh, it is suggested that, textile dyeing industries effluent should meet the discharge quality standards set as per the Environment Conservation Rules (1997). Based on the environmental impact and locations, the industries are classified into four categories (Green, Orange A, Orange B, and Red). Among these, the fabric dyeing and chemical treatment industries in Bangladesh supposed to be under the Red category. Thus, during site clearance the industry must submit an effluent treatment plant (ETP) details to the Department of Environment (DoE) for receiving an environmental clearance certificate. Unfortunately, colour was not included in this existing rule. Considering the adverse effect of the colour in textile wastewater, this study attempted to reduce their levels in effluent.

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Conventional treatment technologies for colour removal include activated sludge, carbon adsorption, ozonation, Fenton reagents, photochemical sonolysis, adsorption, membrane filtration, ion exchange, electro-coagulation, irradiation, biological process and chemical coagulation and flocculation (Verma et al., 2012). Among these, adsorption technology showed best removal performances and this becomes expensive while using commercially available activated carbons. Thus to reduce the treatment cost, there is a need for exploring cheaper substitutes without compromising adsorptive capacity. Because of unburned carbon content and the presence of inorganic (oxide) compounds, fly ash has been investigating as an alternative substrate (Jedidi et al., 2011, Rastogi et al., 2008, Visa et al., 2011, Ahmaruzzaman, 2010).

Fly ash is an industrial by-product, generated during the combustion of coal for energy production. Rastogi *et al.* (2008) could remove approximately 60% of methylene blue from wastewater by using fly ash as an adsorbent and also expected to remove heavy metals(Rastogi et al., 2008). In Bangladesh, coal based power plant in Barapukuria has started generation of fly ash since 2006 and these have generally no use except land filling. The aim of this study is to explore possibilities of using fly ash for a cost effective treatment of textile wastewater from Bangladesh context.

MATERIALS AND METHODS

Among the textile wet processing plants of Bangladesh (i.e., knit dyeing unit, woven dyeing unit, denim plant, printing unit and garments washing units) knit dying industry has identified as one of the highly environment polluting industries. Generated wastewater was collected from a knit dyeing textile industry in Chittagong, Bangladesh for this study (Figure 1). Then the sample wastewater was tested in the environmental laboratory, Chittagong University of Engineering & Technology (CUET).

	Parameters	Studied sample	Standard value for discharging (DoE, 2008)[†]	
			River	Land for irrigation
	Biochemical oxygen demand, BOD$_5$(mg/L)	400	50	100
	Dissolved Oxygen, DO (mg/L)	0.12	4.5-8	4.5-8
	pH	7.98	6-9	6-9
	Total Dissolved Solids, TDS (mg/L)	3570	2100	2100

Figure 1: Conventional wastewater from knit dyeing textile industries

Wastewater sample collected for this study appeared to be highly coloured. Diluting 1% sample with 99% distilled water still showed 100 True Colour Unit (TCU) on Hach colour test disc. As no alternative facilities were available, fly ash was used to test if the reduction of colour reads less than 100 TCU after exposing the water sample to this.

For this pilot study on wastewater treatment, three fly ashes of different compositions were used (Figure 2). Colour of the fly ash could give indication of several properties such as: (a) level of unburned carbon present in fly ash and (b) burning temperature of pulverized feed coal. Among the three types of ashes, the darkest Fly Ash # 2 appeared to contain highest level of activated carbon (Figure 2).Loss-on-ignition (LOI) test were carried out in accordance with the relevant standards EN 196-2 and EN 450-1 respectively (Table 1). As per LOI categories specified in the EN 450-1, Fly Ash#3 was classified in Category A (LOI < 5.0%)

[†] Land Surface Water refers to any pond, tank, water body, water hole, canal, river, spring or estuary; Irrigated Land refers to an appropriately irrigated plantation area of specified crops based on quantity and quality of wastewater.

while Fly Ashes #1 and #2 are beyond the limit. ASTM C618 limits LOI value to 6.0% for both class C and F fly ashes usually considered for use in concrete as supplementary cementitious materials.

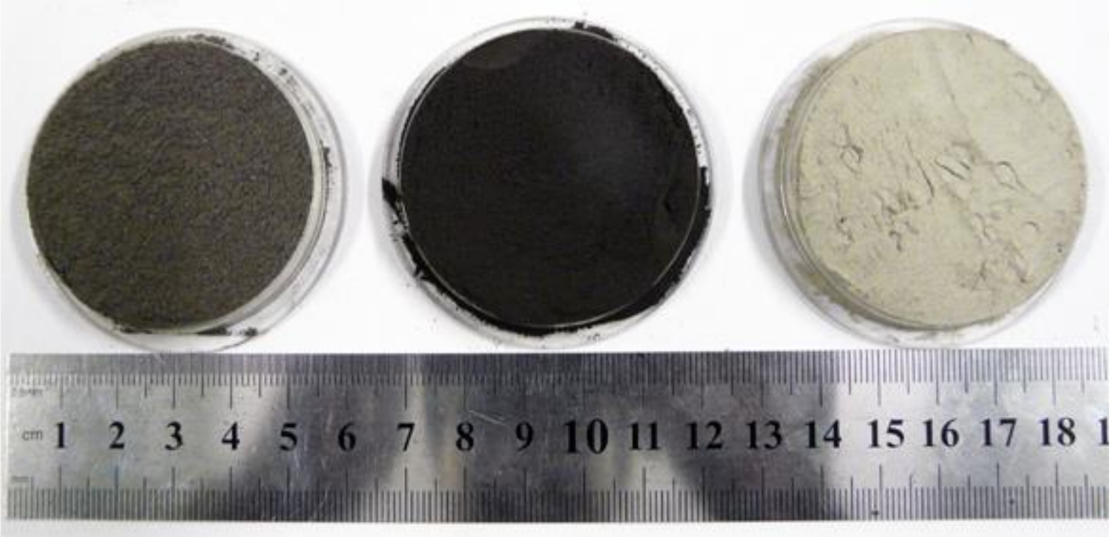

Figure 2: Fly ashes (Left to right reads Fly Ash#1, Fly Ash#2 and Fly Ash#3 respectively) used in laboratory for WWT

Table 1: Physical properties of fly ashes

Properties	Fly Ash #1	Fly Ash #2	Fly Ash #3
Visual observation (Appearance in terms of colour)	Deep grey	Almost black	Whitish grey
[a]LOI, %	24.1	9.1	1.4
Specific surface area, m^2/g	4.9	8.8	0.8

[a] LOI carried out at 975 °C

Initially, different dosages of fly ashes were exposed to collected textile wastewater sample and visual comparative tests were carried out. Fly ashes were preserved in air tight containers throughout the study. Required fly ash transferred to a dry clean beaker and then adding the raw wastewater followed by manually stirring of the beaker. The mixture was then left for a while to allow colour adsorption by fly ash while the contact duration was recorded. Using a Watman 40 filter paper, filtrate of the treated wastewater was collected in a clean tube. Finally, dosages of fly ashes, contact time, and colour of filtrate were recorded (using Hach colour test disc).

RESULTS AND DISCUSSION

Three different dosages in terms of percentages of total sample wastewater were used with each type of fly ash and the contact time were recorded (Table 2). Initially experiments were carried out to evaluate effectiveness of colour removal from the wastewater sample, starting with higher dosages of adsorbent. Initial test with exposing 20% Fly Ash #2 in the sample wastewater gave complete colour removal within minutes. However, in terms of cost effectiveness of the treatment process, both contact time and dosages needs to be adjusted.

Allowing a mean contact time of 15 minutes, Figure 3 compares colour of raw wastewater and exposing it to 5% fly ash of three different types. These show promising outcomes. The most effective performance could even observe visually by using high activated carbon contained Fly Ash#2.

Table 2: Different dosages and the recorded contact timeof fly ashes used to remove colour from textile Wastewater

% of total WW (in volume)	**Sample Wastewater colour (TCU) after using**			**Contact time (minutes)**
	Fly Ash #1	**Fly Ash #2**	**Fly Ash #3**	
2.0	>100	20	>100	60
3.5	>100	15	>100	45
5.0	100	10	100	15

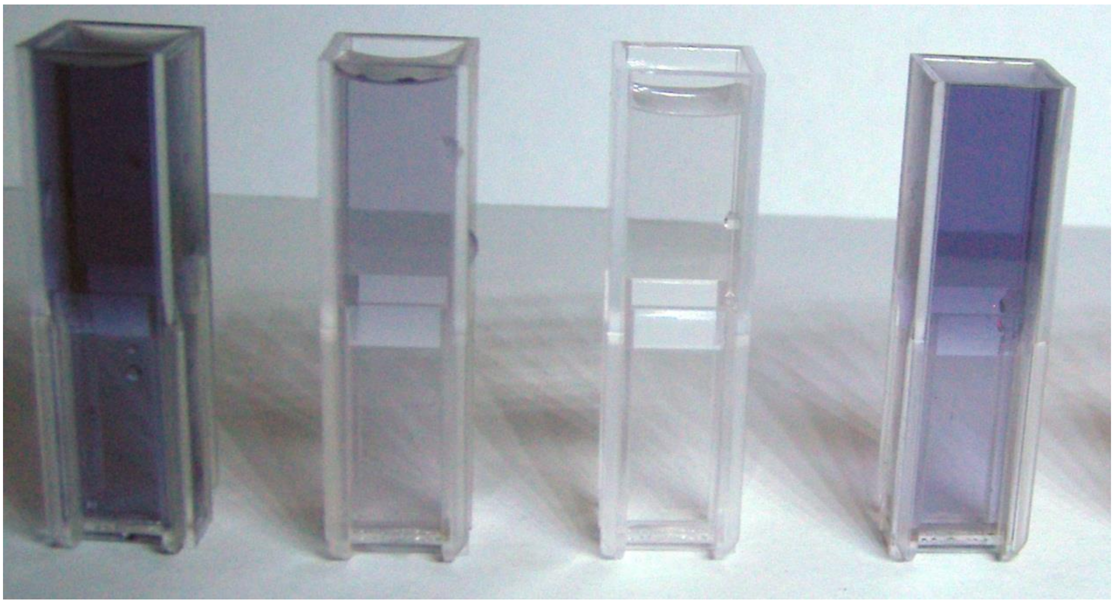

Figure 3: Use of fly as at a 5% dose of the total WW (left to right reads raw WW and colour removed from WW using Fly Ash#1, Fly Ash #2 and Fly Ash #3 respectively)

The LOI and specific surface area (measured by BET N_2 adsorption technology) of the fly ashes under consideration are given in Table 1. The highest LOI value of Fly Ash #1 indicates presence of approximately 3 times unburned carbon then Fly Ash #1. However, their nature of colour adsorption from wastewater did not follow the trend. Recent study (McCarthy et. al., 2012) indicates LOI is not a good indicator of adsorption by fly ash carbon. Rather it was suggested to evaluate performance of adsorbate from its specific surface area, which is also an important parameter for commercial activated carbons. The colour removal results obtained in this study follows the trend of specific surface area and suggests that fly ashes contained higher level of activated carbon could be efficiently used for colour from textile wastewater. The next phase of the study will explore possibilities of leaching of any harmful heavy metal from these fly ashes.

CONCLUSIONS

This study aimed to explore possibilities of using fly ash for dye removal from textile wastewater. Experiments were carried out with various dosages of fly ashes with range of physical properties. In this study, Fly ashes with high activated carbon showed lucrative outcomes, however, details study needs to be carried out to see the seasonal effect on produced fly ashes as well as their details physical properties. In this connection an extended study in is being carried out at CUET. If the future study confirms acceptable/negligible leaching of harmful mineral from fly ashes while in contact with water, this material could be used in WWT as a cost effective adsorbent.

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Study on Energy Recovery from Municipal Solid Waste By Gasification: A Solution of Power Crisis and Enveronmental Pollution

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ABSTRACT

The world is facing rapid growth in energy demand, persistently high energy prices, and a challenge to reduce carbon dioxide emissions from power generation and manufacturing. Many cities are confronted with the problem of how to dispose of large quantities of municipal solid waste (MSW). Currently, landfills are the primary destination of municipal solid waste. However, with landfill tipping fees rising and their proven negative environmental impacts, cleaner and less costly alternatives for municipal waste disposal should be identified and implemented. High temperature energy recovery from MSW, known as waste to-energy (WTE), is one such alternative. Waste-to-energy reduces the amount of materials sent to landfills, prevents air/water contamination, improves recycling rates and lessens the dependence on fossil fuels for power generation. The purpose of this study was to assess MSW gasification technology as an alternative to combustion and landfill and to examine the potentiality of waste-to-energy process.

Key words: *MSW, gasification, landfill, WTE, environment*

INTRODUCTION

Rapid urban expansion, much of it unplanned, generates waste much faster than existing collection and disposal capacity, creating increasingly unserved or underserved populations. The quantity of waste generated per person is steadily increasing, while the quality of that waste is decreasing. Waste generated per person has grown from 0.2-0.5 kilograms a day 30 years ago to 0.5-1.00 kilograms a day today. Meanwhile, the composition of household and business waste has shifted from being almost entirely biodegradable to being much less so, including increasing amounts of high-value recyclables, plastics, and hazardous materials. In developing countries, solid waste management services (primarily transfer and disposal) can consume 20-50 percent of municipal spending, and households and neighborhood associations often pay for primary waste collection services directly [1]. The estimated quantity of Municipal Solid Waste (MSW) generated worldwide is 1.7 – 1.9 billion metric tons [2].In many cases, municipal wastes are not well managed in developing countries, as cities and municipalities cannot cope with the accelerated pace of waste production. Waste collection rates are often lower than 70 per cent in low-income countries. More than 50 per cent of the collected waste is often disposed of through uncontrolled land filling and about 15 per cent is processed through unsafe and informal recycling [3].

There is an overall correlation between the generations of MSW, wealth (Gross Domestic Product, GDP per capita) and urbanization [4]. Future projections estimate that the world's waste production could reach up to 27 billion tons by 2050, a third of which may be generated in Asia, with a significant percentage of that being produced in large economies such as China and India [5]. Currently, landfills are the primary means of MSW disposal taking in most of the residential garbage generated in the world. However, concern about the land requirement and the proven negative environmental impacts have forced to the decision maker to search the alternatives for the municipal solid wastes disposal. Global climate change and its various effects on human life drive current society toward a more sustainable society. Waste is a small contributor to global greenhouse gas (GHG) emissions (<5%) with total emissions of approximately 1,300 $MtCO_2$-eq in 2005, mainly from landfill methane (CH4), followed by wastewater (CH_4 and N_2O); in addition, minor emissions of carbon dioxide (CO_2) result from incineration of waste containing fossil carbon (C) (plastics; synthetic textiles) [6]. Waste-to-energy is one of the technologies for energy recovery from MSW. Waste-to-Energy reduces the amount of materials sent to landfills, can prevent air/water contamination, improves recycling rates and lessens the dependence on fossil fuels for power generation. The two most viable commercial technologies for energy recovery from MSW are combustion and gasification.

WORLDWIDE MUNICIPAL SOLID WASTE GENERATION

Rapid increase in volume and types of solid and hazardous waste as a result of continuous economic growth, urbanization and industrialization, is becoming a burgeoning problem for national and local governments to ensure effective and sustainable management of waste. It is estimated that in 2006 the total amount of municipal solid waste (MSW**)** generated globally reached 2.02 billion tones, representing a 7% annual increase since 2003 (Global Waste Management Market Report 2007). It is further estimated that between 2007 and 2011, global generation of municipal waste will rise by 37.3%, equivalent to roughly 8% increase per year.

The quantity of waste produced in the world has increased considerably over the past decades, especially in wealthy countries. The connection between the national gross domestic product (GDP) and waste generation per capita further supports this argument [7]. It is estimated that the municipal solid waste (MSW) produced globally exceeded 2 billion tons per year at the turn of the millennium, although there is controversy regarding the reliability and consistency of waste data [8]. Naturally, the world population is inherently connected to this increase. The world's population is projected to reach approximately 7.2 billion by 2015 and, by 2025, it is estimated that two-thirds of the world's population will be living in cities [9].

Rising prosperity and the increasing urbanization of the world population could lead to a doubling in the volume of municipal solid waste (MSW) created annually by 2025, according to new research conducted by the World Watch Institute. According to the study, MSW tends to be generated in much higher quantities in wealthier regions of the world. Members of the Organization for Economic Co-operation and Development (OECD), a group of 34 industrialized nations, lead the world in MSW generation, at nearly 1.6 million tonnes per day. By contrast, the report found that sub-Saharan Africa produces less than one eighth as much, some 200,000 tonnes per day. However, the list of top 10 MSW producing countries includes four developing nations (Brazil, China, India, and Mexico) in part because of the size of their urban populations and in part because their city dwellers are prospering and adopting high-consumption lifestyles. Unsurprisingly the U.S. leads the world in MSW output at some 621,000 tonnes per day; China is a relatively close second, at some 521,000 tonnes [10]. Even among the top 10, however, there is a wide range of output, with the U.S. generating nearly seven times more urban refuse than France, in tenth position, does.

Although considerable efforts are being made by many Governments and other entities in tackling waste-related problems, there are still major gaps to be filled in this area. The World Bank estimates that in developing countries, it is common for municipalities to spend 20-50 percent of their available budget on solid waste management (open dumping with open burning is the norm), even though 30-60 percent of all the urban solid wastes remain uncollected and less than 50 percent of the population is served. . In low-income countries, collection alone drains up 80-90 percent of municipal solid waste management budget. In mid-income countries, collection costs 50-80 percent of total budget. In high-income countries, collection only accounts for less than 10 percent of the budget, which allows large funds to be allocated to waste treatment facilities. Upfront community participation in these advanced countries reduces the collection cost and facilitates waste recycling and recovery [11].

In 2010, newspaper/mechanical papers recovery was about 72 percent (7 million tons), and about 58 percent of yard trimmings were recovered. Total MSW generation in 2010 was 250 million tons. Organic materials continue to be the largest component of MSW. Paper and paperboard account for 29 percent and yard trimmings and food scraps account for another 27 percent. Plastics comprise 12 percent; metals make up 9 percent; and rubber, leather, and textiles account for 8 percent. Wood follows at around 6 percent and glass at 5 percent. Other miscellaneous wastes make up approximately 3 percent of the MSW generated in 2010 (Figure 1).

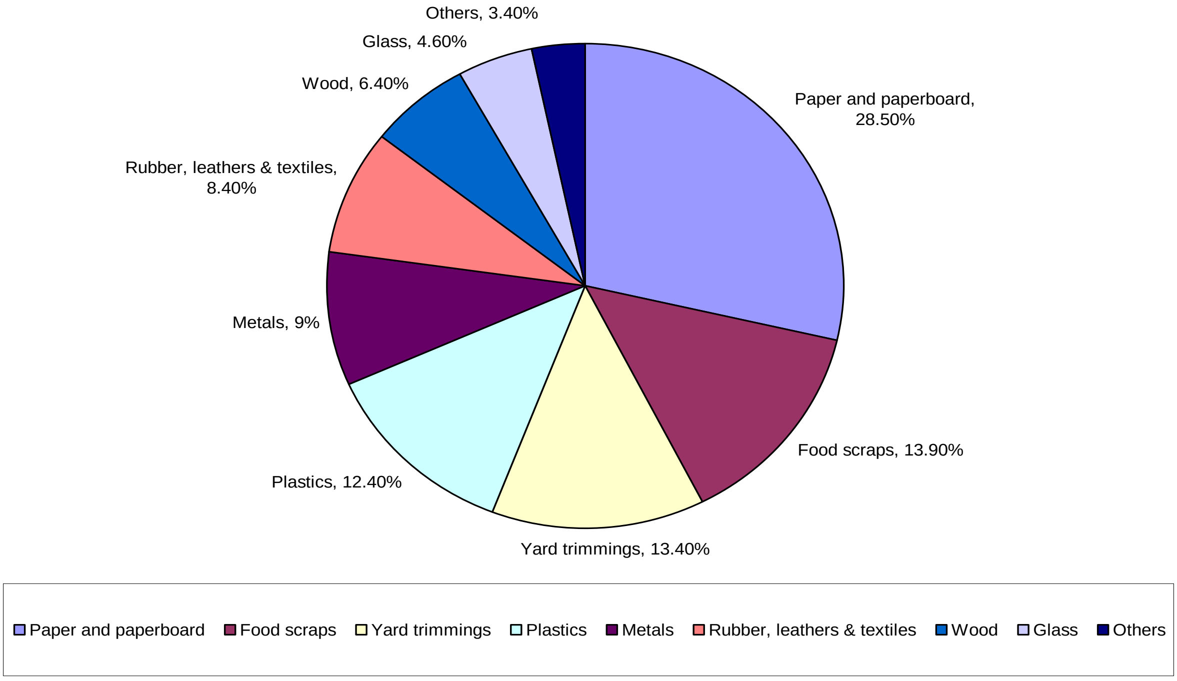

Figure.1. Composition of solid waste

CHARECTERISTICS OF MSW

Solid waste consists of the highly heterogeneous mass of discarded materials such as plastic, polythene, wood, cloth, rubber, bricks, glass and others. Proximate analysis gives information about feedstock suitability in terms of moisture content, volatile matter content and fixed carbon content. Chemical compositions of the solid wastes are identified by the determination of elemental contents in the sample such as Carbon, Nitrogen, Sulfur and Hydrogen. Table 1 shows the proximate and ultimate analysis of different type of solid waste components.

Table 1. Proximate and ultimate analysis of waste components [12, 13, 14]

Waste material	**Waste density (kg/m³)**	**Moisture content (%)**	**Inert residue (%)**	**Calorific value (kj/kg)**	**C (%)**	**H (%)**	**O (%)**	**N (%)**	**S (%)**
Food waste	120-480	50-80	2-8	1540-9900	48	6.4	37.6	2.6	0.4
Garden trimming	60-225	30-80	2-6	478.5-1856.3	47.8	6.0	38	3.4	0.3
Paper	30-130	4-10	6-20	12216-18540	43.5	6.0	44.0	0.3	0.2
Plastic	30-156	1-4	6-20	34900	60	7.2	22.8		
Glass	90-260	1-4	99						
Leather	90-450	8-12	8-20		60	8	11.6	10	0.4
Rubber	90-200	1-4	8-20	16370	78	10		2	
Textile	30-100	6-15	2-4	15770-10600	55	6.6	31.2	4.6	0.15
Wood	156-900	15-40	1-2	14400-17400	49.5	6	42.7	0.2	0.1
Sawdust	250-350	19		15070	49	6			0.1

GASIFICATION AS AN ENERGY RECOVERY OF MSW

Gasification processes involve the reaction of carbonaceous feedstock with an oxygen-containing reagent, usually oxygen, air, steam or carbon dioxide, generally at temperatures in excess of 800°C. It involves the partial oxidation of a substance which implies that oxygen is added but the amounts are not sufficient to allow the fuel to be completely oxidized and full combustion to occur. The process is largely exothermic but some heat may be required to initialize and sustain the gasification process. The main product is a syngas, which contains carbon monoxide, hydrogen and methane. Typically, the gas generated from gasification will have a net calorific value of 4 - 10 MJ/Nm^3.The other main product produced by gasification is a solid residue of non-combustible materials (ash) which contains a relatively low level of carbon. Syngas can be used in a number of different ways, for example:

- Syngas can be burned in a boiler to generate steam which may be used for power generation or industrial heating.
- Syngas can be used as a fuel in a dedicated gas engine.
- Syngas, after reforming, may be suitable for use in a gas turbine
- Syngas can also be used as a chemical feedstock.

Gasification plants, based on syngas production, are relatively small scale, flexible to different inputs and modular development. Producing syngas to serve multiple end-uses could complicate delivery of the plants but it could provide a higher degree of financial security. The most important reason for the growing popularity of thermal processes for the treatment of solid wastes has been the increasing technical, environmental and public dissatisfaction with the performance of conventional incineration processes. MSW is difficult to handle, segregate and feed in a controlled manner to a waste-to-energy facility. MSW has a high tendency to form fused ash deposits on the internal surfaces of furnaces and high temperature reactors, and to form bonded fouling deposits on heat exchanger surfaces. The products of the combustion of MSW are also very aggressive, in that the flue gases are erosive and the relatively high levels of chloride containing species in the flue gases can lead to high rates of metal wastage of heat exchange tube surfaces due to high temperature corrosion.

While evaluating gasification or other thermal technologies, the degree of pre-processing required in conversion of MSW into a suitable feed material is a major criterion. Unsorted MSW is not suitable for most thermal technologies because of its varying composition and size of some of its constituent materials. It may also contain undesirable materials which can play havoc with the process or emission control systems. The main steps involved in pre-processing of MSW include manual and mechanical separation or sorting, shredding, grinding, blending with other materials, drying and pelletization. The purpose of pre-processing is to produce a feed material with consistent physical characteristics and chemical properties. Pre-processing operations are also designed to produce a material that can be safely handled, transported and stored.

Types of gasifiers for MSW treatment

Gasification technology is selected on the basis of available fuel quality, capacity range, and gas quality conditions. The main reactors used for gasification of MSW are fixed beds and fluidized beds. Larger capacity gasifiers are preferable for treatment of MSW because they allow for variable fuel feed, uniform process temperatures due to highly turbulent flow through the bed, good interaction between gases and solids, and high levels of carbon conversion. Table 2 shows the thermal capacity ranges for the main gasifier designs.

Table 2: Thermal capacity of different gasifier designs [14]

Downdraft	1 KW – 1MW
Updraft	1.1MW – 12MW
fluidized-bed gasifiers	1 MW – 50MW
Cross-draft gasifiers	10 MW – 200MW

Fixed Beds

Fixed bed gasifiers typically have a grate to support the feed material and maintain a stationary reaction zone. They are relatively easy to design and operate, and are therefore useful for small and medium scale power and thermal energy uses. The two primary types of fixed bed gasifiers are updraft and downdraft.

In a downdraft gasifier, air is introduced into a downward flowing packed bed or solid fuel stream and gas is drawn off at the bottom. The air/oxygen and fuel enter the reaction zone from above decomposing the combustion gases and burning most of the tars. Downdraft gasifiers are not ideal for waste treatment because they typically require a low ash fuel such as wood, to avoid clogging.

In an updraft gasifier, the fuel is also fed at the top of the gasifier but the airflow is in the upward direction. As the fuel flows downward through the vessel it dries, pyrolyzes, gasifies and combusts. The main use of updraft gasifiers has been with direct use of the gas in a closely coupled boiler or furnace. Because the gas leaves this gasifier at relatively low temperatures, the process has a high thermal efficiency and, as a result, wet MSW containing 50% moisture can be gasified without any pre-drying of the waste.

Slagging fixed bed gasifier, which is high-pressure and oxygen-injected, has commercial potential for gasifying MSW. In theory, the high temperatures crack all tars and other volatiles into non-condensable, light gases. Also under these conditions, the ash becomes molten and is tapped out, as is done in iron blast furnaces.

Fluidized Beds

Fluidized beds are an attractive proposition for the gasification of MSW. In a fluidized bed boiler, a stream of gas (typically air or steam) is passed upward through a bed of solid fuel and material (such as coarse sand or limestone). The gas acts as the fluidizing medium and also provides the oxidant for combustion and tar cracking. Waste is introduced either on top of the bed through a feed chute or into the bed through an auger. Fluidized-beds have the advantage of extremely good mixing and high heat transfer, resulting in very uniform bed conditions and efficient reactions. Fluidized bed technology is more suitable for generators with capacities greater than 10 MW because it can be used with different fuels, requires relatively compact combustion chambers and allows for good operational control. The two main types of fluidized beds for power generation are bubbling and circulating fluidized beds.

In a Bubbling Fluidized Bed (BFB), the gas velocity must be high enough so that the solid particles, comprising thebed material, are lifted, thus expanding the bed and causing it to bubble like a liquid. A bubbling fluidized bed reactor typically has a cylindrical or rectangular chamber designed so that contact between the gas and solids facilitates drying and size reduction (attrition). As waste isintroduced into the bed, most of the organics vaporize pyrolytically and are partially combusted in the bed. Typical desired operating temperatures range from 900° to 1000 °C[15].

A circulating fluidized bed (CFB) is differentiated from a bubbling fluid bed in that there is no distinct separation between the dense solids zone and the dilute solids zone. The capacity to process different feedstock with varying compositions and moisture contents is a major advantage in such systems.

ENVIRONMENTAL BENEFITS OF GASIFICATION OVER COMBUSTION AND LANDFILL

There are numerous solid waste gasification facilities operating or under construction around the world. Gasification has several advantages over traditional combustion processes for MSW treatment It takes place in a low oxygen environment that limits the formation of dioxins and of large quantities of SOx and NOx. Furthermore, it requires just a fraction of the stoichiometric amount of oxygen necessary for combustion. As a result, the volume of process gas is low, requiring smaller and less expensive gas cleaning equipment. The lower gas volume also means a higher partial pressure of contaminants in the off-gas, which favours more complete adsorption and particulate capture. Finally, gasification generates a fuel gas that can be integrated with combined cycle turbines, reciprocating engines and, potentially, with fuel cells that convert fuel energy to electricity more efficiently than conventional steam boilers.

Environmental benefits of MSW gasification compared to landfill of solid may include:

The public health, safety, and environmental concerns fall into three categories: subsurface migration, surface emissions/air pollution, and odor nuisance.

Subsurface Migration

Subsurface migration is the underground movement of landfill gas from landfills to other areas within the landfill property or outside the landfill property. (Note: Most subsurface migration occurs at older, unlined landfills because there is minimal barrier for lateral migration. The Resource Conservation and Recovery Act began requiring all new or expanded landfills to be lined as of October 9, 1993. This requirement decreases the likelihood of subsurface migration.) Since landfill gas contains approximately 50 percent methane (a potentially explosive gas) it is possible for landfill gas to travel underground, accumulate in enclosed structures, and ignite. There have been incidences of subsurface migration causing fires and explosions on both landfill property and private property.

Surface Emissions

Possibly the biggest health and environmental concerns are related to the uncontrolled surface emissions of landfill gas into the air. As previously mentioned, landfill gas contains carbon dioxide, methane, VOC, HAP, and odorous compounds that can adversely affect public health and the environment. For example, carbon dioxide and methane are greenhouse gases that contribute to global climate change. Methane is of particular concern because it is 21 times more effective at trapping heat in the atmosphere than carbon dioxide. Emissions of VOC contribute to ground-level ozone formation (smog). Ozone is capable of reducing or damaging vegetation growth as well as causing respiratory problems in humans. Finally, exposure to HAP can cause a variety of health problems, such as cancerous illnesses, respiratory irritation, and central nervous system damage. Thermal treatment of NMOC (including HAP and VOC) and methane through flaring or combustion in an engine, turbine, boiler, or other device greatly reduces the emission of these compounds.

Odors

The final concern related to uncontrolled landfill gas emissions is their unpleasant odor. Compounds found in landfill gas are associated with strong, pungent odors. These smells can be transmitted off-site to nearby homes and business. Unpleasant odors can lower the quality of life for individuals that live near landfills and potentially reduce local property values.

Environmental benefits of MSW gasification compared to combustion of solid biomass may include:

Reduced carbon emissions by efficiency improvements

Gasification has potential to increase energy efficiency compared to combustion of biomass in a steam cycle. These carbon emission reductions may be tradable in carbon offset markets. Significant production of biochar reduces energy efficiency, if the char is not reburned. But biochar offers other environmental advantages that can more than make up for its energy efficiency penalty.

Reduced carbon emissions by closing the carbon cycle and carbon sequestration

Both fossil fuels and biomass release $CO_2$when they burn. The carbon released when burning fossil fuels originates from oil reserves, not from the atmosphere. Hence, fossil fuels are carbon positive in that they add new carbon dioxide to the atmosphere. In contrast, combustion of biomass, taken by it, is carbon neutral because the carbon released was first absorbed from the atmosphere by the biomass as it grew. In other words, the carbon cycle is closed. Combustion of biomass may still be carbon positive overall if fossil fuels are used in their production and transportation. Use of biomass has the potential of being carbon negative if, in using or producing it, carbon is stored in a form that is not released to the atmosphere.

Reduced fertilizer use and runoff in biochar-amended soils

Biochar as a soil amendment significantly increases the efficiency of and reduces the need for traditional chemical fertilizers, while greatly enhancing crop yields. Production and transportation of chemical fertilizers is fossil fuel intensive and so reducing their use reduces associated carbon emissions. Moreover, char-amended soils have shown 50% to 80% reductions in nitrous oxide emissions, reduced runoff of phosphorus into surface waters, and reduced leaching of nitrogen into groundwater.

Reduced NOx emissions

The product gas will generally have low NOx concentrations because gasification temperatures are not high enough to produce NOx in significant quantities. However, when the product gas is burned in a boiler, turbine or engine, NOx will be produced as it is in most combustion systems & with all fuels. Nevertheless, it is easier to control the combustion of a gaseous fuel than the combustion of a solid fuel. Better control of combustion provides to reduce NOx formation. Some of the environmental benefits of the gasifire:

- Gasification plants produce significantly lower quantities of criteria air pollutants.
- Gasification can reduce the environmental impact of waste disposal because it can use waste products as feedstock's generating valuable products from materials that would otherwise be disposed as wastes.
- Gasification plants use significantly less water and can be designed so they recycle their process water, discharging none into the surrounding environment.
- CO_2 can be captured from an industrial gasification plant using commercially proven technologies.
- Gasification offers the cleanest, most efficient means of producing electricity from coal and the lowest cost option for capturing CO_2 from power generation [16].

CONCLUSION

A solution to the waste problems confronted by municipalities requires a strategy that integrates several technologies including, waste reduction, recycling, landfilling and waste-to-energy. Waste-to-energy, which converts the non-recyclable and combustible portion of the waste to electricity, reduces the amount of materials sent to landfills, prevents air/water contamination, improves recycling rates and lessens the dependence on fossil fuels for power generation. Another area that would increase the viability of waste gasification is the improvement of waste sorting and pre-treatment methods. Preparation of a homogenous RDF remains one of the most difficult tasks in thermochemical conversion of solid waste. It involves a large amount of mechanical processing and close supervision, which greatly impact operating costs and can account for as much as 40% of the total plant capital costs. If shredding and sorting of the waste can be made simpler and more effective, gasification would become even more advantageous. Similarly, waste gasification will be most successful in communities where there is good recycling practice. A better job of recycling glass and food wastes by city residents will improve the gasification reactions. Converting Municipal Solid Waste (MSW) to energy has the environmental advantages of reducing the number of landfills, preventing water/air contamination, and lessening the dependence on oil and other fossil fuels for power generation. Gasification is a WTE technology that can be cost competitive with combustion and offers the potential for superior environmental performance. However, before it can be considered to be a clear-cut solution for waste disposal in large municipalities, its long-term reliability must be demonstrated.

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Monitoring Of a Low Cost Ceramic Filter by Observing the Chemical Parameters for Actual Wastewater Treatment

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ABSTRACT

In this study a low cost and simple type ceramic membrane was innovated for concurrent wastewater treatment and reuse. As the MBR processes are still costly in terms of price of membrane, maintenance cost and energy consumption. Locally available and cheap materials (clay soil and rice bran) were used as ingredients to decline the membrane cost in this study. Mixing ratio of the ingredients were 80% of clay soil and 20 % of rice bran on weight basis. The ceramic membrane was submerged inside reactor to formulate as ceramic membrane bioreactor (CMBR). For this research study 3 numbers of lab-scale Ceramic Membrane Bio-Reactors (CMBRs) were set up. The average Turbidity Removal efficiency of all Reactors was 95.34%. The result shows that about 88.45% Color removal was achieved by the system. The average SS removal efficiency was 95.25% by all Reactors which demonstrates that the removal of SS was very efficient by CMBR. About 81.55% COD removal was achieved by Ceramic Membrane Bio-reactor. The CMBR has great potential in removing biodegrading organic pollutants from wastewater. As the Ceramic Membrane was made by locally available materials the technology was inexpensive. Therefore, the technology is suitable and can be adapted in developing countries for wastewater treatment and reuse.

INTRODUCTION

Due to diminishing water supplies and increasing population, wastewater reclamation is becoming necessary throughout the world to conserve natural water resources used for drinking water supply.The demand for water is increasing with the increasing industrialization, urbanization and the diversification of lifestyles (Kim and Cho 1993). In addition to this, there have been problems relating to the increased amount of wastewater also, since aside from a minor quantity, most consumed water is transformed into wastewater. Wastewater is water that has been used and must be treated before it is released into another water body, so that it does not cause further pollution of water sources (SDWF Report 2007) or any water that is no longer wanted, as no further benefits can be derived out of it, is termed as wastewater (Hasan and Nakajima 2010). Generally there are four types of wastewater such as Domestic wastewater, Municipal wastewater, Industrial wastewater and Storm wastewater. Domestic wastewater can be divided into Black waster and Grey water. Generally greywater is divided in four greywater categories based on its origin: bathroom, laundry, kitchen and mixed origin (Morel 2005). Greywater includes all household wastewater except toilet waste. It can be a valuable water resource, and an increasing number of householders are recycling greywater for a variety of purposes. However, care must be taken with this practice as it can carry health and environmental risks (Environmental Health Unit Report 2003). In Bangladesh, the greywater situation in terms of treatment and recycling is the most horrible. Most of the people think that greywater is not so much polluted like blackwater and so not only rural parts but also in urban area there is no treatment options for greywater. And greywater recycling is still unbelievable. Generally different scenarios are prevailed in urban and rural areas of Bangladesh about greywater (Hasan and Nakajima 2010). Among all the technologies, membrane technology is found effective and hence gets the incomparable popularity in recent years for wastewater treatment and reuse. Membrane bioreactor (MBR) technology is characterized as a combination of biological wastewater treatment (WWT) and membrane separation, by which biomass can retained in the system without conventional gravity sedimentation (Itokawa 2009). The Membrane Bioreactor is a simple, but very effective combination of the activated sludge treatment process and the membrane filtration process (Operator Notebook Report 2001). The MBR process is an emerging advanced wastewater treatment technology that has been successfully applied at an ever increasing number of locations around the world (Chapman et al. 2003). Membrane bioreactor (MBR) is an improvement of the 100-year old CASP (Conventional Activated Sludge Processes), where the traditional secondary clarifier is replaced by a membrane unit for the separation of treated water from the mixed solution in the bioreactor (Xing et al. 2000).

MATERIALS AND METHODS

Manufacture of Ceramic Membrane Module

For making the Ceramic Membrane Bio-Reactor cheap, easily and locally available materials (rice bran and clay soil) were chosen for wastewater treatment. About 80% clay soil and 20% rice bran was used for manufacture of ceramic membrane. Locally collected clay soil samples were dried and grind by hammer. The dried and grind clay soil was sieved by 0.5 mm mesh. The rice burn was also dried and sieved by 1 mm mesh. Mixing ratio of the ingredients were 80% of clay soil and 20 % of rice bran on weight basis. This mixing proportion of the materials was selected by quantifying pore volume, pore size, compressive strength and filtration efficiency of several ceramic bars prepared with different ratios of the ingredients. Details about the selection of ingredients proportion were described in the next section. To make the dough manually, dried ingredients were mixed homogeneously and then sufficient amount of water was used. Water of 400-500 ml was used with the dry homogeneous-mixed ingredients of 800 gm for making one membrane module. In the end, a hollow cylindrical shape was manually made with the dough as 10 cm height with 10 cm outer diameter and 6 cm inner diameter with one side opened (Figure 1). To make preferred shape, a wooden dice and PVC pipe of 10cm ht with 10cm outer diameter cut vertically in symmetrical were used. The membrane was then kept for 24 hrs for natural dry at room temperature, then oven dried at 105°C for 24 hrs and finally burnt in a muffle furnace in the laboratory. The temperature in muffle furnace was increased from room temperature to 900°C and kept this temperature for 2 hrs. After that Temperature was decreased gradually as the firing was terminated. After the termination of firing the membrane modules were kept into the kiln for overnight and take out from the kiln in next morning.

Experimental Set-up

For this research study 3 numbers of lab-scale Ceramic Membrane Bio-Reactors (CMBRs) were set up as named R-1, R-2, and R-3. All the three Reactors were as cylindrical column shaped with 14 cm inner diameter and made by PVC pipe. The inner diameter of all the Reactors was same as 14 cm and heights was same as about 140 cm. In each reactor, Membrane Module was directly submerged inside the Reactor. The Membrane was placed on a PVC plate and was made water tight by using Silica glue to prevent infiltration of water, tilting and floating. The PVC plate with Membrane was then placed at the end of PVC pipe. The Bio-Reactors were filled with raw wastewater. The wastewater used in this study was actual wastewater rather than synthetic Wastewater. The wastewater was collected from a drain near Dr. M. A Rashid Hall in KUET. The raw wastewater used as feed composed of effluent water from septic tank and bathing water from the residential Hall. The concentration of various ingredients of raw wastewater was different. The intermittent flow of influent was applied in this study. The objective was to measure influent and effluent characteristics of the raw wastewater. For this purpose various water quality parameter was determined to measure the performance of Ceramic Membrane Bio-Reactor. The wastewater was fed into each Reactor from the feed tank by peristaltic pump. The CMBRs were aerated from the beneath of membrane module through a diffuser, so that rising air bubbles can provide the membrane surface with more shear stress, which is effective for removing attached sludge out of membrane, and to mix the mixed liquor in the Reactor and also to maintain an aerobic environment for the normal growth of activated sludge. Intermittent aeration system was set up as 4 hr aeration per day by using the blower. The three Reactors R-1, R-2 and R-3 were under aerobic condition. Permeate from the Reactors was collected during the aeration period through outlet by gravitational pressure. The water was sampled every three days per week. The parameters analyzed were the level of Turbidity, Color, TS, TDS, SS, Fe and COD in the Bio-Reactor and Flow rate. Figure 3 shows the schematic view of the system, displays photographs of the membrane media position in the Reactor and the CMBR under continuous operation in the laboratory. The CMBR systems were monitored by measurement of permeate flux, pH and DO. Sludge retention time (SRT) was infinitive as there was no sludge wastage except for sampling during the operation. The Run time of Reactor-1, Reactor-2 and Reactor-3 were 215 days. Within this time period Reactor-1 and Reactor-2 were clogged after 129 days from 29-12-10 to 04-05-11. Reactor-3 was clogged after 65 days and after cleaning it was clogged again at 129 days. A cake layer formation was found on all Ceramic Membrane surfaces because of the deposition of the floe sludge. The membrane was then cleaned physically by removing the accumulated sludge with the help of water, knife and soft spongy brush. There was no need of chemical washing or change of the membrane.

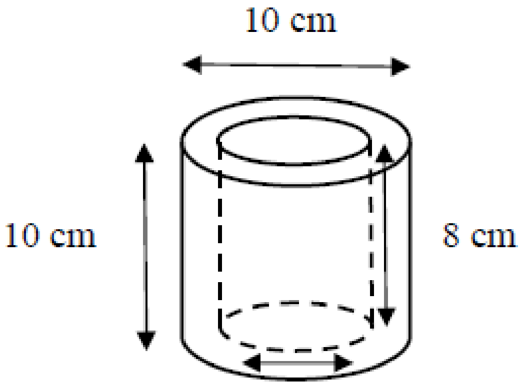

Figure 1: Dimension of Ceramic Membrane

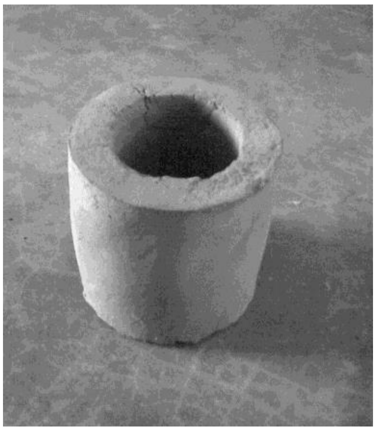

Figure 2: Ceramic Membrane after burn

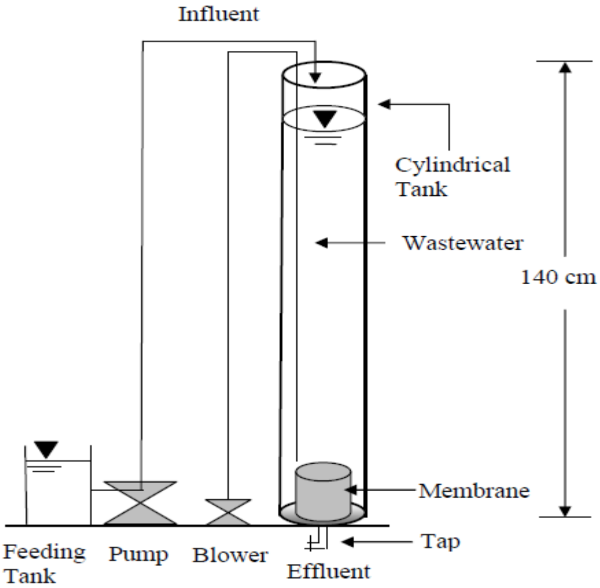

Figure 3: Schematic representation of the system

RESULTS AND DISCUSSION

Operating Parameters (pH & DO)

From laboratory test it was found that the average DO in influent wastewater was 3.47 mg/l. The average effluent DO by Reactor-1, Ractor-2 and Reactor-3 were 5.42 mg/l, 5.42 mg/l and 5.47 mg/l, respectively. During the accumulation of effluent water into the bucket it has been found that the effluent water was exposed to air and the value of DO increase because of aeration. The standard value of DO is 4.5 to 8 for discharging the wastewater into land water, public sewer and on irrigated land. The average effluent DO by all Reactors was 5.44 mg/l which is within the standard value. In water supply, pH is very important as the organism involved in treatment processes operate within a certain pH range. The pH value in influent was within the range of 6.68 to 8.80 and the average pH value in influent was 7.97. The average pH value in Reactor-1, Reactor-2 and Reactor-3 were 7.99, 7.92 and 7.91 respectively. The pH of all Reactors was stable within the range of 7-8 as shown in Figure 4.

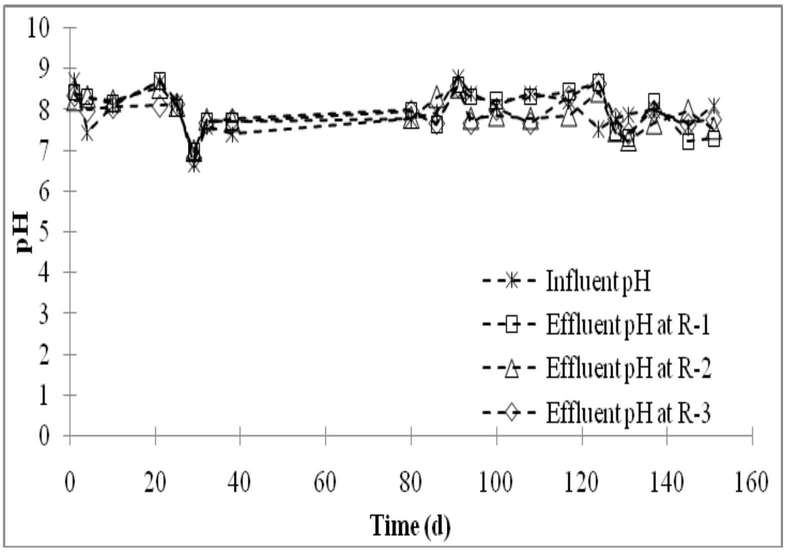

Figure 4: Range of pH in CMBR

COD Removal

Disposal of wastewater containing high COD to receiving water bodies might cause oxygen depletion that will have harmful effects to living resources like fishes, or eventually make the environment anaerobic. Therefore, its removal is given more focus in any wastewater treatment facility. Through the study it was found that the average COD of influent wastewater was 572.91 mg/l with minimum and maximum value of 92 mg/l and 1568 mg/l respectively. After treatment by CMBR the effluent showed very low COD as the average value of COD in effluent of Reactor-1, Reactor-2 and Reactor-3 were 68.40 mg/l, 84.91 mg/l and 78.36 mg/l respectively during the operation period. The average effluent COD value by all Reactors was 77.22 mg/l.

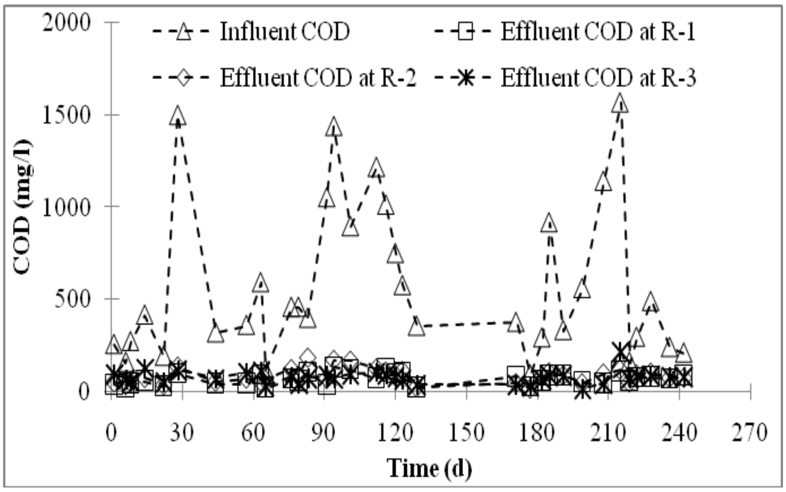

Figure 5: COD in Influent and Effluent

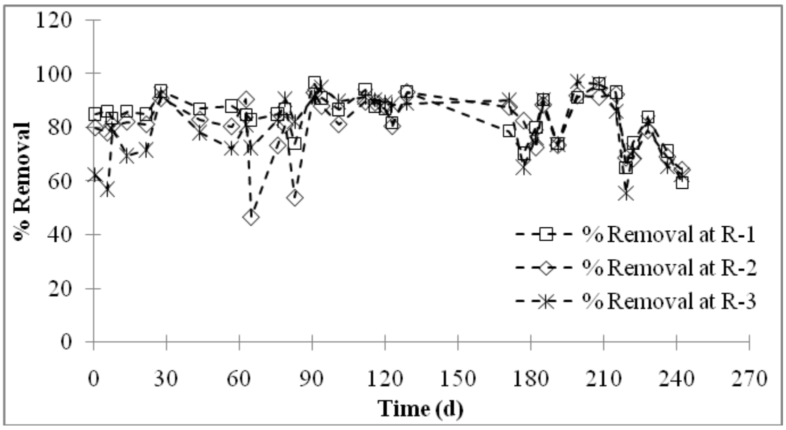

Figure 6: COD Removal Efficiency

Fe Removal

In this study, it was found that significant Fe removal was achieved by CMBR. The average concentration of Iron in influent wastewater was 0.2 mg/l which was very low because the wastewater used as influent in CMBR was surface water. As the concentration of Iron in Influent water was very low, the data collection was carried out up to 129 days. The average concentration of Iron (Fe) in effluent water was 0.02 mg/l with minimum and maximum value of 0.0 mg/l and 0.06 mg/l respectively. The concentration of Fe in influent and effluent water is shown in Figure 7.The Iron removal efficiency of Reactor-1, Reactor-2 and Reactor-3 were 93.99%, 88.98% and 95.1% respectively and the average removal efficiency of all Reactors was 92.69%. Figure 8 shows the Fe removal efficiency of all Reactors.

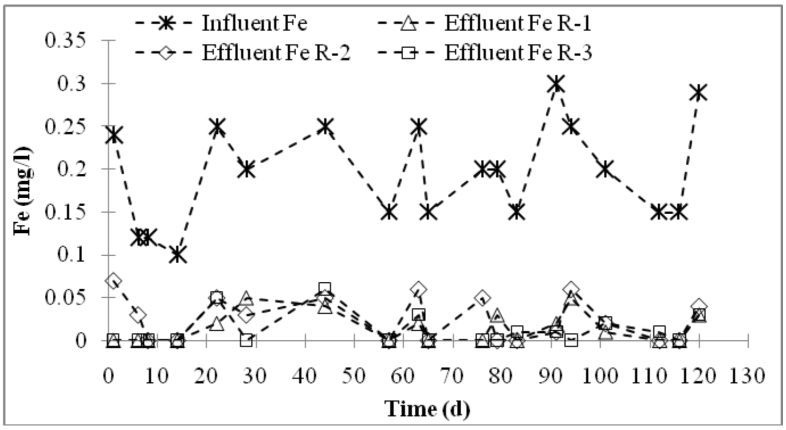

Figure 7: Concentration of Iron in Influent and Effluent

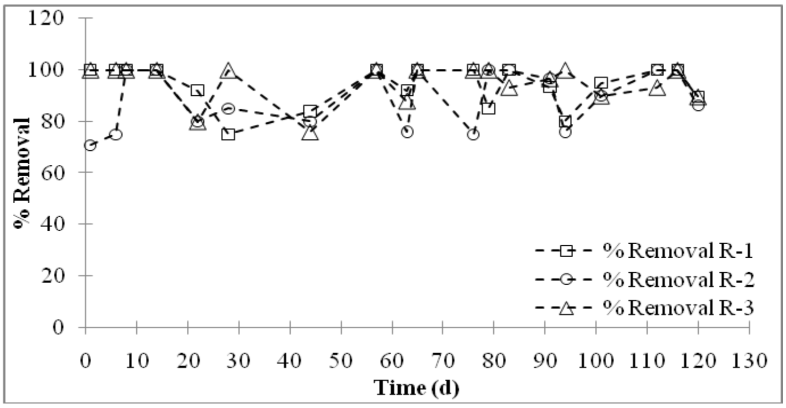

Figure 8: Iron Removal Efficiency

BOD Removal

From previous study it has been found that BOD concentration of synthetic wastewater used during the operation period was 5000 mg/l but effluent BOD was lower than 5 mg/l which meant, about 99.9% of BOD was removed (Hasan and Nakajima 2010). In this study actual wastewater was used as feed and it has been found that among six samples, the average BOD concentration of raw wastewater used during the operation period was 5.33 mg/l but average effluent BOD was about 0.96 mg/l which means that about 82% of BOD removal was achieved by CMBR. Overall, it can be concluded that the CMBR has great potential in removing biodegrading organic pollutants from wastewater.

Chloride Test

Through laboratory test considerable amount of Chloride was found in influent and effluent wastewater which means that there was salinity in wastewater. Salinity or Chloride removal efficiency by filtration process is poor. For example from Chloride test, it was found that influent chloride was 1050 mg/l where as effluent chloride was 790 mg/l which means that about 1711.5 mg/l NaCl was present in influent water and about 1287.7 mg/l NaCl was present in effluent water and other kinds of salt like $CaCl_2$ or KCl was also present in wastewater.

Other Findings

The total amount of wastewater used for all Reactors is about 550.8 liters (183.6 Liters for each Reactor). After 129 days of operation about 113 liters of wastewater was used for each Reactor. After 242 days of operation about 70.2 liters of water was used for each Reactor.

Table 1 Overall condition of Reactor-3 before Clogging

Days	No of Sample	Water Used (Liters)	Total Solids in Influent (Kg)
At 65 days	10	54	0.21
At 129 days	11	60	0.20
At 242 days	13	70	0.21

From the above Table it has been found that the amount of Total solids was about 200 gm when the Reactor-3 was clogged. So clogging depends on the amount of Total Solids used as feed.
By considering similar situation of Reactor-1 and Reactor-2 before and after Clogging, it has been found that Reactor-1 and Reactor-2 was clogged and cleaned at 129 days. After 242 days when all Reactors were cleaned again then the Flow rate of R-1 and R-2 were 5 ml/min and 5 ml/min respectively. Table 4-9: shows the situation of Reactor-1 and Reactor-2 at similar flow rate before and after clogging.

Table 2: Overall condition of Reactor-1 and Reactor-3 at similar flow rate (about 5 ml/min)

No of days	Total Sample	Flow rate at R-1 (ml/min)	Flow Rate at R-2 (ml/min)	Water Used (L)	TS in Influent (Kg)
Before Clogging at 91 days	14	5	4	75.6	0.25
After clogging at 242 days	13	5	5	70.2	0.2

From the study it has been found that the flow rate of Reactor-1 and Reactor-2 at 91 days was about 5 ml/min (before clogging at 129 days) and the flow rate of Reactor-1 and Reactor-2 at 242 days was also about 5 ml/min (after clogging at 129 days). At 91 days the Total Solids in Influent used as feed was 250 gm and at 242 days the Total Solids in Influent used as feed was 200 gm. So the amount of Total Solids was very similar when the flow rate was about 5 ml/min for both Reactors before and after clogging. The total amount of water used as feed before and after clogging were 75 liter and 70.2 liter respectively (when the flow rate was about 5 ml/min). So Flow rate and Clogging of CMBR depend on the amount of Total Solids used as feed.

CONCLUSION

A low cost and simple type ceramic membrane was innovated for concurrent wastewater treatment and reuse especially for wastewater treatment in this study. This simple type CMBR process was investigated from the laboratory experiments and it can be concluded that: Removal efficiency of organic matters in terms of COD was very much satisfactory in the case of wastewater treatment. About 81.55% of COD removal was achieved by Ceramic Membrane Bio-reactor. From the results, it can be concluded that the anoxic-aerobic CMBR has great potential in removing biodegrading organic pollutants from wastewater. This indicates that the Ceramic Membrane is able to retain the organic content from wastewater. Iron removed significantly by CMBR. The average Iron removal efficiency of all Reactors was 92.69%. The WHO guideline suggests that the concentration of Iron should be less than 0.3 mg/l in drinking water. The average concentration of Iron in effluent water was found 0.02 mg/l by CMBR which is well below even drinking water standard. So the Ceramic Membrane Bio-Reactor is very efficient to remove Iron from wastewater. Physical cleaning of the membrane was much simple and it was easy to remove the cake layer to reclaim the membrane. The run time of CMBR was very good. The longer and maximum run time of CMBR was 129 days. The quality of effluent water was excellent as the effluent water was clear colored and odor- free. It was found that high removal efficiency of organic content was obtained that could be made it suitable for wastewater reuse. As the Ceramic Membrane was made by locally available materials the technology was inexpensive. Therefore the technology is suitable and can be adapted in developing countries for wastewater treatment and reuse. The total cost for setup of Ceramic Membrane Bio-reactor in the laboratory was about 16500 Taka only.

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Microbiological management for vermicomposting of a slowly degradable organic waste

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Abstract

In recent years, vermicomposting biotechnology is emerging as an effective biotechnology for recycling different organic wastes using epigeic earthworms which degrade different kinds of organic materials with the help of large numbers of intestinal microorganisms resulting in better nutrient status of the produced vermicomposts, as compared to those prepared by traditional composting systems. However, most of the organic wastes used for composting usually contain some relatively resistant components, especially cellulose which limits humification of these wastes. Introduction of some cellulose degrading microorganisms have been known to hasten the decomposition of such components. Since information pertaining to use of such microbes with regard to vermicomposting is meager, an attempt was made in this study to assess the efficiency of two cellulose degrading fungi viz. *Trichodermaviridae* and *Pleurotussajarcasu* in degrading a slowly decomposing organic waste,. sugarcane trash. This waste was treated with different combinations of Trichoderma and Pleurotus and the effects of such inoculations on composting of the organic waste were assessed with regard to periodic changes in pH, microbial biomass carbon, microbial respiration, cation exchange capacity, easily mineralisable nitrogen and also midway occurrence of these microbes in the substrates and earthworm intestines under different treatments. Inoculations of these microorganisms were found to increase the microbiological activities in the earthworm intestines and also in the composting substrates. These, in turn, helped to enhance the pace of vermicomposting of such relatively resistant organic wastes. Of the two microorganisms, *Trichodermaviridae* appeared to be the more efficient for the purpose of vermicomposting.

Keywords: Organic waste recycling; Vermicomposting; Resistant components; Microbial management.

Introduction

In recent years, vermicomposting has emerged as an easily adoptable biotechnology for recycling wide ranges of organic wastes. During the course of vermicomposting, earthworm gut microorganisms breakdown the organic wastes to form vermicompost (Senapati, 1993). However, in many cases, these wastes contain some components like cellulose, lignin etc which are slowly degradable or comparatively resistant to decomposition .Commonly occurring microorganisms, in general, find it difficult to decompose such organic compounds. Under this situation, introduction of some specific microorganisms with proven efficiency of degrading these resistant organic compounds may appear to be effective.Gaur (2006), while discussing the benefits of using such microorganisms for accelerating decomposition of resistant organic compounds, mentioned several organisms for such purpose. However, information on use efficiency of these microorganisms under vermicomposting system is still very meager. In the present investigation, the efficiency of two cellulose decomposing microorganisms viz. *Trichoderma viridae* (Gaur, 2006) and *Pleurotus sajarcasu* (Zechendorf, 2005) in composting sugar cane trash, a slowly degradable organic waste, in absence and presence of epigeic earthworms has been studied.

Materials and Methods

For the purpose of the study, a slow decomposing organic waste viz. sugarcane trash, which has high cellulose content, was used. These wastes were taken in earthen pots, mixed with cow dung at 1: 1(W/W) ratio and treated with Trichoderma and Pleurotus using the following treatment combinations :

i) Organic wastes (OW)
ii) OW + 10 nos. of earthworm (EW) per kg waste
iii) OW + 10 EW + *Trichoderma viridae* (TV) @ 1%
iv) OW + 10 EW + *Pleurotus sajarcasu* (PS) @ 1%
v) OW + 10 EW + TV @ 0.5% + PS @ 0.5%

Each of the treatments was replicated 4 times under randomized block design.
Microorganisms procured from market contained $10x10^5$ no. of organism count per g of material. The microbes were inoculated after reduction of initial flush of heat in the composting medium. This was followed by introduction of earthworms and maintenance of 40-50% moisture in the substrates covering a period of 60 days of incubation.

Periodic samplings were done at 15 days intervals and the samples were analyzed for, pH (1:2), microbial biomass carbon (Vance *et al*. (1987), microbial respiration (Alef and Nannipieri, 1995), cation exchange capacity (Harada and Inoko ,1980) and easily mineralisable nitrogen (Subbiah and Asija ,1954) for assessing the microbiological activity as well as the rate of decomposition.

Results and Discussion

Sugarcane trash, the residual material after extraction of sugar juice; contains high concentrations of cellulolytic materials and is ,therefore, found to be highly resistant to decomposition. In this study, the effects of vermicomposting in absence and presence of different cellulose degrading fungi have been assessed.

The pH values of the composting materials ranged between 6.55 and 6.98 under different treatments during the period of inoculation (Table- 1). The variations were found to be insignificant between the treatments. The pH values were considered to remain under congenial range for promoting the growths of the earthworms and the decomposing microorganisms. Use of different treatments did not exert any significant effect on pH values of the substrates.

Microbiological activity,as indicated by microbial biomass carbon (MBC) value, was found to be the lowest in the control series (Table- 1) due to obvious reasons. Introduction of earthworms helped to improve the MBC values due to the effects of earthworm gut microorganisms. Inoculation of *Trichoderma viridae* and *Pluerotus sajarcasu* singly and also in combination, to the vermicomposting system increased the MBC values considerably and the increments were found to be statistically significant over the only vermicomposting series in most of the cases. In all the treatments, MBC values were higher during the initial period of study and showed a declining trend as the composting assumed maturation. Increased abundance of microorganisms during early stages of decomposition and it’s gradual reduction with the completion of composting has been discussed by Brady (1980) and many others.

Almost similar was the situation with microbial respiration under different treatments (table-1). Respiration values were higher during the initial period of the study and declined gradually at the later phases. Such gradual reduction of microbial activity with the completion of decomposition is well documented (Nannipieri*et al.*,1990). Introduction of cellulose degrading microorganisms in the vermicomposting treatments helped to increase the microbial respiration values over the only vermicomposting series and the highest increments were obtained in the treatment with 50% Trichoderma *Sp.* and 50% Pleurotus S*p.*. On the other hand, there was practically no variation in basal respiration values between Trichoderma S*p.*and Pleurotus *Sp* .treatments when applied at 100% rates. This indicates that degradation of more resistant components of these organic wastes were more effective when both these organisms were used together.

Table 1.pH, MBC and microbial respiration values in different treatments during the period of incubation

Treatments	**pH**				**MBC(µgg^{-1})**				**Microbial respiration(mg. CO$_2$ g^{-1} hr^{-1} at 25^o C)**			
	Days after incubation											
	15	**30**	**45**	**60**	**15**	**30**	**45**	**60**	**15**	**30**	**45**	**60**
O.W.(control)	6.55	6.60	6.68	6.77	178.41	160.33	151.86	147.33	2.00	1.78	0.98	0.64
O.W. + E.W	6.75	6.73	6.81	6.93	223.06	201.96	154.06	171.73	2.13	1.86	1.05	0.67
O.W. + E.W + T$_{100\%}$	6.77	6.54	6.72	6.90	234.04	232.89	172.96	176.77	2.36	2.04	1.50	0.76
O.W. + E.W + P$_{100\%}$	6.83	6.50	6.96	6.85	242.82	254.66	191.41	189.32	2.38	2.04	1.52	0.74
O.W. + E.W + T$_{50\%}$ + P$_{50\%}$	6.98	6.78	6.95	6.90	237.76	246.09	211.83	189.32	2.41	2.62	1.72	0.90
C. D.	0.50	0.41	0.44	0.48	12.328	14.635	12.595	14.275	0.143	0.269	0.190	0.115

O.W.= Organic wastes, E. W. = Earthworm, T= *Trichodermaviridae*, P = *Pleurotussajarcasu*

These variations in microbiological population as well as activities under different treatments influenced the humification of the organic wastes considerably. This behaviour was reflected in variations in cation exchange capacity (CEC) of the substrates at different periods of incubation under various treatments (Table-2). Importance of CEC as an important indicator of composting has been discussed by Harada and Inoko (1980) and others. The CEC values, in general, showed an increasing trend with the period of incubation under all the treatments. This was obviously owing to increased opening of exchange sites with the course of composing. High CEC values of humic substances have been reported by Kononova *et al*(1966) and others. The property showed numerically higher values with introduction of the microorganisms and the highest CEC values were observed when both Trichoderma *Sp.*and Pleurotus *Sp.* were used together. As discussed earlier, this treatment showed comparatively higher MBC and basal respiration values also. Such increased microbial activity helped the composting process to proceed at faster rate resulting in more complete humification. This was reflected in higher CEC values under these treatments. However, the variations in CEC values among the microbiological treatments were not found to be statistically significant always.

Table -2.CEC and available nitrogen in different treatments during the period of incubation

	CEC(C. mol. (P^+) kg^{-1}**)**				Available Nitrogen (mg kg-1)			
	15	**30**	**45**	**60**	**15**	**30**	**45**	**60**
O.W.(control)	155.85	171.83	187.61	189.53	487.41	618.82	711.25	811.25
O.W. + E.W	179.75	178.57	197.57	201.57	534.36	867.57	1008.55	1025.57
O.W. + E.W + $T_{100\%}$	180.61	197.67	207.17	211.92	688.71	1026.05	1035.25	1052.34
O.W. + E.W + $P_{100\%}$	187.63	192.25	197.25	199.25	606.26	839.36	985.53	1006.97
O.W. + E.W + $T_{50\%}$ + $P_{50\%}$	173.31	195.73	199.35	203.74	607.45	859.03	868.33	974.08
C. D.	18.72	20.34	23.489	21.90	53.04	69.75	57.53	68.64

O.W.= Organic wastes, E. W. = Earthworm, T= *Trichoderma viridae*, P = *Pleurotus sajarcasu*

Since nitrogen forms a major component of organic materials which is gradually transformed into available form with the mineralization of organic matter, the easily mineralizable form of organic matter is likely to indicate the rate of decomposition of any organic waste. In the present study, therefore, occurrence of easily mineralisable nitrogen was used as an indicator of the rate of decomposition of the organic wastes. The values of easily mineralisable nitrogen showed an increasing trend with the period of incubation under all the treatments, obviously owing to consistent mineralization of the organic form of nitrogen during the courses of decomposition of the wastes. Adoption of vermicomposting helped to increase the availability of nitrogen due to enhanced rate of decomposition. Introduction of microorganisms in the composting systems increased the occurrence of easily mineralizable form of nitrogen further due to more intense microbiological activities. In general, use of Trichoderma *Sp.@* 100% maintained higher amount of easily minerablizable nitrogen over the other two treatments. This was in contrary to the general observations that the treatment with 50% Trichoderma *Sp.* + 50% Pleurotus *Sp.* treatment showed comparatively higher rates of microbial activity, as were evident by MBC and basal respiration values. This behaviour may be due to occurrence of very wide C : N ratio in these organic wastes which contained higher amount of cellulolytic and lignin like compounds.Higher microbial activity in the above mentioned treatment increased the rate of microbial decomposition of the wastes, as was evident by increased CEC values (Table -2). However, at the same time, release of nitrogen to mineral forms became limited due to immobilization of the nitrogen under wide C : N values and also intensified microbial activities. For preventing immobilization and encouraging mineralization of nitrogen during the composting of such slowly degradable organic wastes, use of nitrogen fixing microbes or application of small amountof nitrogenous fertilizer, as suggested by Gaur and Singh (1995), may appear to be effective .

Conclusion

The results of the study indicated that use of cellulose degrading microorganisms may be considered as an effective proposition for enhancing the rate of vermicomposting of cellulose rich slowly degrading organic waste like sugarcane trash. While both the cellulose degrading fungus viz. Trichoderma *Sp* and 50% Pleurotus *Sp* appeared to be effective in this regard, combined use of both the microorganisms was found to be more effective in composting such waste materials. However, immobilization of nitrogen may be a probable problem during such composting of cellulose rich waste material with high C: N ratio. Further studies may be carried out in this regard.

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Use of Compost for Imoroving Coastal Salt Affected Soil to Sustain Crop Yields

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ABSTRACT

This study investigated the physical and chemical properties of agricultural soil in the south-west coastal region of Bangladesh. Most of the soil falls under saline categories and textures are varied from silty clay to silty clay loam; containing intolerable levels of soluble salts (ECe > 4). The possible impacts on plants are individual ionic toxicity, less availability of water to the plant roots and deficient plant essential nutrients. The fundamental characteristics of these soils have a major effect on the structure of soil stratum. The arrangement of soil particles is critical in reducing soil permeability, hydraulic conductivity and infiltration rate due to swelling, flocculation and dispersion of clays and slaking of soil aggregates. These soils possess a number of challenges that include poor physical and chemical properties that affecting the growth of most crops. However, the application of composted municipal solid waste could be a promising alternative to alleviate the adverse effects. The focal aim for the application of compost in such soil is to improving the properties of soil through high cation exchange capacity and offset nutrient depletion. Nevertheless, it can be a great opportunity to use the high organic matter containing compost as reclamation of salt-affected soils for sustaining coastal agricultural productivity and constitute low cost soil recovery.

INTRODUCTION

Soil salinization and nutrients deficient in agricultural land are a severe problem throughout the world and around 20% of the world’s cultivated land and 50% of cropland are affected (Flowers et al., 1995). Globally, its effect on total cultivated land, approximately (23%) is saline and (37%) is sodic at different degrees. In Bangladesh, the coastal area about 30% of the country's total cultivable land has been significantly contributing to the agricultural sector from the very beginning; but unfortunately now a day it also affected by salinity/sodicity. The use of low quality water and traditional agricultural practice is deteriorating this problem day by day (Lakhdar et al., 2008). The excessive amount of soluble salts adversely affects the soil physical and chemical properties as well as it can induce specific-ion effects by increasing the concentration of ions with an inhibitory effect on biological metabolism. Tejada and Gonzalez, (2006) has stated a relationship that the soil structural stability and bulk density are decreased as increasing electrical conductivity. In coastal areas, clayey textural soil reveals excessive amount of exchangeable sodium and high pH that favors swelling and dispersion of clays as well as slaking of soil aggregates through the decrease soil permeability, available water capacity and infiltration rate (Lauchli and Epstein,1990). The extensive evaporation rate tends to accumulate salts in the upper soil profile; especially it occurs where leaching facilities is insufficient

(Isabelo and Jack, 1993). However, the adverse effect of soil salinity on plant growth and productivity varies with the type of plant being grown and salinity stress.

The rational use of soil fertilizers, using good quality water and appropriate cultural practices are amelioration technique to sustain allowable properties of salt affected soil (Grattan and Oster, 2003). The common reclamation techniques are: deep ploughing, leaching, sub soiling, sanding, and profile inversion are used as physical amelioration; the application of gypsum, calcium chloride, and limestone for chemical amelioration; and treatment with electric current used as electro-reclamation (Raychev et al., 2001). Therefore, regarding all of the above techniques have limitations only physical and biological amendment may effective in such affected soil.

The application of organic matter conditioner (compost) in agricultural land is a common practice from the last decades to regeneration and enhancement of soil fertility (Melero et al., 2007). The effects of organic fertilization on chemical and biochemical properties of an agricultural soil exhibited increases in quantity and quality of total organic carbon, nitrogen and phosphorus nutrients, microbial substrate and enzymatic activities. In the same way, the use of such amendment have two principal beneficial effects on reclamation of saline soil: improvement of soil structure and permeability thus enhancing salt leaching, reducing surface evaporation and inhibition of salt accumulation in surface soils, and release of carbon dioxide during respiration and decomposition (Raychev et al., 2001). However, absurd or low quality uses of compost may causes potential threat thereby release of organic and inorganic pollutant in the soil which can adversely affect organisms and ecosystems (Cai et al., 2007). Hence, qualitative compost may fulfill all of these constraints and sustain agricultural productivity as well as soil ecosystem. This paper focuses the effectiveness of compost in coastal agricultural lands where salinity level is not very high but often decline yields of sensitive crops.

METERIALS AND METHODS

Description of study area

The study area covers the South-west coastal plain of Bangladesh at the apex of the Bay of Bengal and covers approximately 12,212 km^2 (Figure 1). The area lies between 21^{0} 30˝ and 23^{0} 15˝ North and 89^{0} 00˝ and 90^{0} 00˝ east and includes the world's largest continuous mangrove forest, the Sunderban, estuarine swamplands and numerous rivers, canals and their tributaries. The terrain is relatively flat; soil of this area is alluvial floodplain, the average land elevation is 2-3 m above from mean sea level. The annual average temperature and rainfall is 35.5^{0}C and 1710 mm respectively. Total cultivable land area was estimated around 508650 hectares and the main crops are: paddy, jute, potato and vegetables.

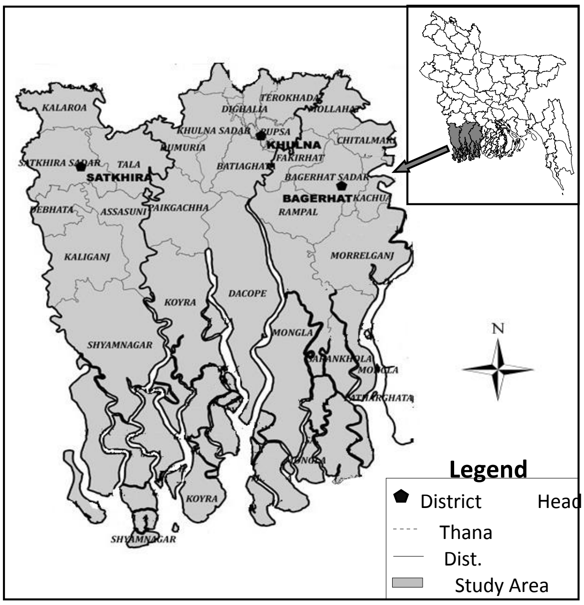

Figure 1 Location of study area

Soil sampling

The soil samples were collected from eighteen locations of agricultural land at depth intervals of 0–10, 20–30 and 40-50 cm. The samples were collected adopting with coring technique (Prawit et al., 2002) and conserved in air dried plastic bags. After collection, the samples were dried in air and then grinded and passed through 1-mm sieve.

Laboratory analysis

The salinity related variables (chloride, soluble sulfate, and soluble cations) were determined using the methods suggested by Rump and Krist (1992). After extracting the soil samples with distilled water (1/5, w/v), chloride (Cl^-), soluble sulfate (SO_4^{2-}), and soluble cations (Ca, Mg, Na, K) were determined respectively by $AgNO_3$ titration, Turbidimetric method, and atomic adsorption spectrometry. The soil nutrients (NO_3-N and PO_4-P) were determined using Spectrophotometer (DR4000, Hach, USA). The other parameters: pH (1/2.5) and EC (1/5) were determined using the methods suggested by Rowell (1994). The pH was measured by pH-meter in suspension with distilled water (1/2.5, w/v). The EC was measured by EC-meter in the supernatant suspension with distilled water (1/5, w/v) and converted to 25^0C. For particle size distribution, air-dried soil samples were dried again in oven and passed through No. 200 sieve and analyzed by hydrometer method. The electrical conductivity of saturated extracts (ECe) was calculated as follows:

$$ECe(dS/m) = EC_{1:5}(dS/m) * Multiplier.factor \qquad (1)$$

Where, multiplier factor was calculated according to soil texture (table 1) (Slavich and Petterson, 1993).

Table 1 Multiplier factors for calculate ECe (dS/m)

Soil texture	Factors
Sand, loamy sand, clayey sand	23
Sandy loam, fine sandy loam, light sandy clay loam	14
Loam, fine sandy loam, silty loam, sandy clay loam	9.5
Clay loam, silty clay loam, fine sandy clay loam, sandy clay, silty clay, light clay	8.6
Light medium clay	8.6
Medium clay	7.5
Heavy clay	5.8

For the characterization of soil salinity and sodicity, the common index SAR was calculated as follows:

$$SAR = \frac{[Na^+]}{\sqrt{\frac{[Ca^{2+}]+[Mg^{2+}]}{2}}} \qquad (2)$$

Where, $[Na^+]$, $[Ca^{2+}]$ and $[Mg^{2+}]$ are concentration of water soluble ions (meq/L) in soil saturation extract.

RESULTS AND DISCUSSION

Salinity stress and compost amendment

Salt affected soils are very diverse in characteristics and exhibit a combination of the salinity stresses. This soil appraises two major salinity stresses to crops, which are the concentration of total soluble salts and the quantity of exchangeable Na^+. The soluble salt ions [Cl^{1-}, SO_4^{2-}, soluble cations (Ca, Mg, Na, K)] are relating to the potential for salt-induced drought stress and consequent osmotic adjustments internally in the plant. According to US Salinity Laboratory, most of sample characterized as saline soil (Table 2) and the range, mean and standard deviation of these parameters were found significant variation (Table 4). In this study area, all soils are deficient in plant essential nutrients of nitrogen (N), phosphorus (P), and potassium (K), because; salinity can affect forms and dynamics of this nutrient in soil (Dominguez et al., 2001) and also limits in soil fertility (Lakhdar et al., 2008).

Table 2 Classification of Salt-Affected Soils by U.S. Salinity Laboratory

Soil class	Total salinity	Sodium permeability hazard	
	ECe (dS/m)	SAR	pH
Saline	>4	<12	<8.5
Sodic	<4	>12	>8.5
Saline-Sodic	>4	>12	<8.5

In addition, clay textural soils are not capable to retain much P by itself due to negative charge (Table 3). In such case, compost amendments firmly can use to provide nutrients (N and P and K) to rebuild soil physical and chemical properties. The soil profile was observed much variation of all parameters except Ca^{+2}. The high concentration was observed at depth (0-10) and (40-50), but relatively low at depth (20-30) for most of all parameters (Table 4). The physical properties (texture) were observed in between silty clay to silty clay loam and the percent of clay content increasing with increase the depth of soil profile (Table 3). Therefore, this soil profile may reflect insufficient leaching facilities due to poor physical properties. The physical property of soil is strongly dependent on presence of Na^+ content, which observed significant amount in this area. Sodium is not a plant nutrient and therefore it is not necessary for plant growth. High levels of Na^+ are damaging to soil aggregation and plant growth. Moreover, It acts as highly dispersive agent resulting breakup of soil aggregates (Debuysera et al., 2004) and the exchangeable Na^+ in the soil solution at the exchange sites contribute to repulsive charges that disperse clays particles (Figure 2a). However, the addition of organic matter assist to glues the tiny soil particles together into larger water stable aggregates, promotes the flocculation of clay minerals, increasing bio-pores spaces which increase soil air circulation necessary for growth of plants and microorganisms (McConnell et al., 1993) (Fig. 2b). It also act as enrichment of the exchange complex in Ca^{2+} and Mg^{2+} , since it could decrease the proportion of Na^+ in the exchange complex, improving soil physical properties.

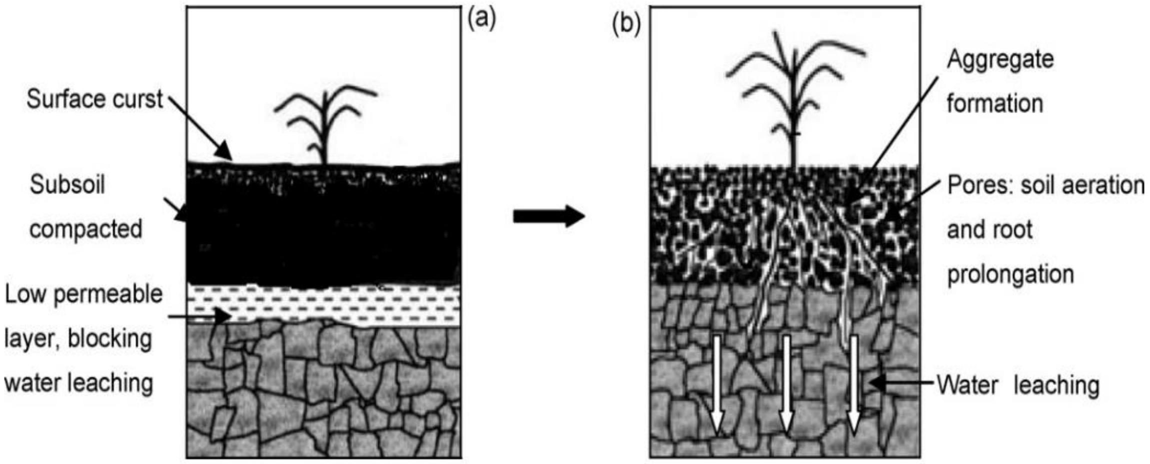

Figure 2 Schematic representation of the soil before (a) and after (b) organic amendment addition (Lakhdar et al., 2009)

Table 3 Ranges of observed particle size distribution and soil texture

Silt (%)	Clay (%)	Sand (%)	Texture
(44 - 48)	(36 - 40)	(12 – 16)	Silty clay, silty clay loam

Table 4 Soluble salts and nutrients in the studied coastal soils of Bangladesh

Parameters (unit)	Depth (cm)	Min	Max	Mean	SD
pH	0-10	6.79	7.13	6.97	0.13
	20-30	5.96	7.30	6.90	0.63
	40-50	7.11	7.62	7.37	0.36
Cl^- (mg 100 g^{-1} soil)	0-10	24.99	194.94	95.97	62.47
	20-30	34.99	49.98	44.99	7.07
	40-50	49.98	76.23	63.11	18.56
Ec_e (dS/m)	0-10	1.70	10.89	6.66	3.68
	20-30	1.53	3.34	2.77	0.85
	40-50	3.35	3.86	3.60	0.36
Soluble SO_4^{2-} (mg 100 g^{-1} soil)	0-10	20.00	86.00	48.80	29.55
	20-30	16.00	42.00	27.00	12.27
	40-50	30.00	32.50	31.25	1.77
Soluble K (mg 100 g^{-1} soil)	0-10	0.63	3.97	2.09	1.23
	20-30	0.56	3.43	1.61	1.27
	40-50	1.09	2.24	1.66	0.82
Soluble Na (mg 100 g^{-1} soil)	0-10	6.05	104.65	62.92	41.11
	20-30	4.82	46.50	31.50	18.99
	40-50	31.69	40.54	36.12	6.26
Soluble Ca (mg 100 g^{-1} soil)	0-10	12.69	133.33	62.49	57.70
	20-30	9.25	144.36	95.69	60.63
	40-50	6.94	112.31	59.63	74.51
Soluble Mg (mg 100 g^{-1} soil)	0-10	5.19	15.56	8.58	4.40
	20-30	0.12	5.32	3.45	2.29
	40-50	2.54	7.78	5.16	3.70
NO_3-N (mg 100 g^{-1} soil)	0-10	0.20	1.40	0.70	0.41
	20-30	0.80	1.60	1.16	0.36
	40-50	0.60	1.00	0.87	0.16
PO_4-P (mg 100 g^{-1} soil)	0-10	0.08	1.48	0.52	0.51
	20-30	0.46	5.06	1.85	1.96
	40-50	0.06	1.88	1.03	0.71
SAR	0-10	0.46	8.68	3.39	3.17
	20-30	0.61	1.46	1.17	0.40
	40-50	1.41	3.69	2.55	1.61

Compost quality and functions in soil improvement

Composting is a transformation from raw unstable or biodegradable waste materials to stable and mature end product as biological process under favorable conditions. Hence, compost has been using in agricultural land to improve soil quality; however it is necessary to evaluate its quality. However, the criteria's to evaluate compost quality are very difficult; because it depends on the original organic matter as well as process of composting. Several researchers have studied on physical, chemical and biological properties of compost and its suitability to salt affected soil (Table 5).

Table 5 Recommended and/or typical range of compost quality for use in crops land

Properties	Unit	Recommended range	Typical range	References
pH	-	6-7.5	-	Leaon M., (1995)
EC (1:1 soil/ water)	dS/m	<2.5	-	Leaon M., (1995)
Bulk density	Mg m^{-3}	0.22–0.74	-	He et al., (1995)
Organic matter (DW)	%	>30	-	EPA waste-licencing system
C/N	-	<25	-	EPA waste-licencing system
Faecal coliform	MPN/g	<1000	-	Herity L., (2003)
TN	%	>1	1.0–3.0	Barker A.V., (1997)
K (DW)	%	-	0.6-1.7	-
MSW -P (DW)	g/kg	-	5-35	Hargreaves et al., (2008)
Ca (DW)	%	-	1-4	-
Mg (DW)	%	-	0.2-0.4	-

The pH interacts directly in the availability of plant nutrients and compost apparently functioned to buffer the pH of saline and alkaline soil. When compost is applied to soil, pH often decreases due to effects of nitrification and increases due to acidification process in acid soil (Walker et al., 2004). Total soluble salts should measure in compost to which plant roots will be exposed. Compost bulk density measurement is expected as it helps in soil aggregation. The compost amended soil can increase the CEC from 20 to 70% of the original CEC (Havlin et al., 1999). The temperature is significant for the microbial activity during biodegradation process. Hachicha et al., (2008) stated an increase of temperature indicates an intensive microbial activity as higher degradation rates. It is well established that compost contains sufficient organic matter and frequent application of MSW compost consistently increased soil organic matter content and soil C/N ratio (Figure 3).

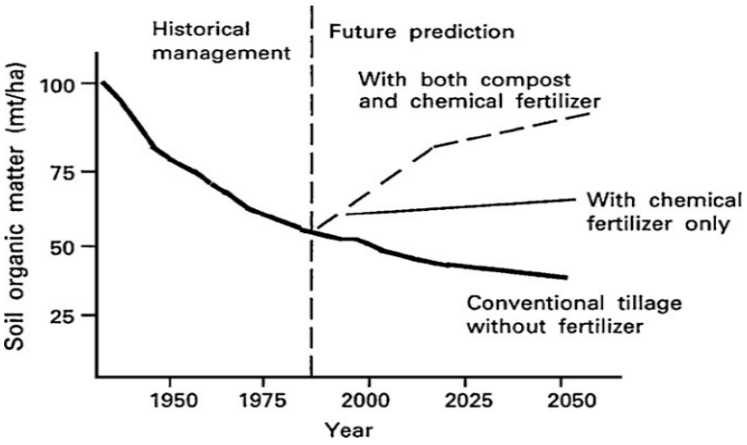

Figure 3 Changes in soil organic matter content (mt/ha) calculated in Taiwan under different soil management systems with long term application of composts or fertilizers (Chen et al., 1998).

Organic matter increases the availability of substrates to soil microorganisms and it regulates the soil ecosystem. The adequate C/N ratio shows a positive effect in microbial and enzymatic activities when organic matter added to saline soils (Tejada et al., 2006). The compost may contain pathogenic organisms and to reduce these health risks, the compost shall confirm up to acceptable limit. The decomposition of organic matter release gradually plant available phosphorus. Compost contain calcium and magnesium which act as bases when they exist as oxides, hydroxides and carbonates forms to soil and may counteract soil acidification (Fricke and Vogtmann,1994).

CONCLUSIONS AND RECOMMENDATION

Compost uses does not completely solve the salinity problem in agricultural land. However, the use of compost on saline soils still improves soil physico-chemical properties, microbial biomass and growth of plants. In addition, long term application of this soil conditioner could restoration soil physical properties and thus favorable to leach of dissolve salt ions. Therefore, we proposed that compost can be used in shallow rooted agricultural land with low to medium salinity.

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Investigation of Solid Waste Management and Surrounding Groundwater Quality at Rajbandh Site at Khulna City of Bangladesh

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ABSTRACT

This paper presents the outline of a demonstration project that aims to develop a safe and sustainable system for the management of Municipal Solid Waste (MSW) at disposal site in Bangladesh through the practical application of improved sanitary landfill. Khulna, the third largest city of Bangladesh and situated at the Southwest region of the country, Rajbandh in Khulna is considered as the case study area. This study also critically identified (Waste Safe 2005) the present status and constraints of MSW management of the study areas and proposed an approach to solve this problem putting priorities on some specific areas. To ensure a clean, hygiene and environmental-friendly city, the city authority is looking for a safe and sustainable solution for the appropriate management of solid wastes. The objective of this paper is to evaluate the groundwater contamination due to the landfill leachate and Proposals for modification of waste management practices at Rajbandh dumping site and final recommendation. Groundwater samples were collected one, two and three km radius around the Rajbandh dumping site. The parameters measured were pH, BOD, COD, Conductivity, Alkalinity, Hardness and Total Dissolved Solid (TDS). The value of other water quality parameter Biochemical oxygen demand (BOD),Chemical Oxygen Demand (COD), Electrical Conductivity, Alkalinity, Hardness and Total Dissolved Solid (TDS), is increasing respectively three, two and one km radius.

INTRODUCTION

Bangladesh is a densely populated country in the third world facing myriads of problems with the growth of population. The increased population leads to the growth of urban areas and slums which, in turn, generating a huge volume of waste. A large proportion of the waste is not properly managed and dumped in unplanned sites that are creating severe environmental hazards. Human beings right from the time of birth are encircled or surrounded by peoples, animals, plants, land soil, other physical objectives. All this are part of human environment. Environment thus the sum of all-social, biological, physical and chemical factors which compose the surround of human begins. The world commission on Environment and development (WCED) known as burnt land commission defined as “where we live” and sustainable develop as “The development that meets the needs of present without compromising the ability of future generations to meet there own needs. Inadequate management of solid waste is an obvious cause for degradation of the environment in most cities of the third world”. Where intense human are concentrated, such as in urban areas, appropriate and safe solid waste management of (MSW) are of utmost importance to allow healthy living conditions for the population. MSW is considered one of the major global environmental problems, especially in LDACs and most important solid waste because of its nature and impact on our community; it consists of hazardous and non-hazardous wastes (Zurbrugg 2002). It is a fact that solid waste composition differs from one community to another community to their culture and socio-economic level. However, solving inadequate management of solid waste in general is very challenging because of its heterogeneous nature. On the other hand, solving the problem in urban area of developing countries is more challenging because of two factors that is

a) Low socio-economic level of the majority of population and their lack of awareness of scope of problem as well as

b) Lack of suitable of a suitable technology platform needed to face problem

Open dumps are the oldest and the most common way of disposing of solid wastes, and although in recent years thousands have been closed, many are still being used. In many cases, they are located wherever land is available, without regard to safety, health hazard and aesthetic degradation. The waste is often piled as high as equipment allows. In some instances, the refuse was ignited and allowed to burn. In others, the refuse was periodically leveled and compacted. As a general rule, open dumps tend to create a nuisance by being unsightly, breeding pests, creating a health hazard, polluting the air and sometimes polluting groundwater and surface water. Landfill is an engineered waste disposal site facility with specific pollution control technologies designed to minimize potential impacts. Landfills are usually either placed above ground or contained within quarries and pits Landfills are sources of groundwater and soil pollution due to the production of leachate and its migration through refuse. Municipal solid waste did not pose a significant problem until human established settlements near landfill.

Prior to that, the types and quantities of waste were readily degraded or consumed by animals or naturally degraded without causing significant impact to the environment. Municipal solid waste did not pose a significant problem until human established settlements near landfill. Prior to that, the types and quantities of waste were readily degraded or consumed by animals or naturally degraded without causing significant impact to the environment and groundwater quality if leachate is discharging into these water bodies. Groundwater is that portion of subsurface water which occupies the part of the ground that is fully saturated and flows into a hole under pressure greater than atmospheric pressure. Groundwater occurs in geological formations known as aquifer. An aquifer (gravel/ sand) may be defined as a geologic formation that contains sufficient permeable materials to yield significant quantities of water to wells and springs. This implies the ability of the formation to store and transmit water. Groundwater is an important source of drinking water for humankind. It contains over 90% of the fresh water resources and is an important reserve of good quality water.

Overview of Study Area

Rajbandh Trenching Ground Site-It is at present the only official dumping site, 25 acres in area situated at a distance of about 10 kms to the west of KCC Headquarter. Field investigators were stationed at the dumping sites to collect information on the nos. of trucks, capacity measurement of trucks, nature of solid waste and the origin of the waste. Data were collected on 16th, 17th and 18th June 2005. Four field investigators were engaged there from 6.00AM to 12:00 midnight for actual waste transportation. 16-19 waste carrying trucks were employed per day at Rajbandh Trenching Ground. Attempts were made to determine the weights of truck with load and without load to determine the actual load. Unfortunately, no scale is at present available in the Khulna city to carry out the measurement. Bulk/packing density was therefore, measured separately by making a container of 8 cft. volume (2ft.x 2ft .x 2ft.). As shown in Table 1 average bulk/packing density is found to be 15.5 kg/cft. i.e. 547.31 kg/m3. There are 4 types of trucks with nameplate capacities of 7-tonne, 6-tonne, 5-tonne and 1.5- tone. The volume of each type is shown in Table 2. Multiplying the volume with bulk density (547.31 kg/m3), the full load weight for 7-tonne, 6-tonne, 5-tonne and 1.5-tonne trucks are 6.228 tones, 4.926 tones, 3.284 tones and 0.766 tones respectively. When a truck is over loaded, the full load weight is multiplied by 1.2; a truck is overloaded to 1.5 times the height of the truck at the center i.e. conservatively about 20% overloading for a rectangular truck. Under loaded trucks carry less than full load and is not less than 50% of the full load. An average of 0.75 is assumed as load factor and the actual weight is obtained by multiplying the full load weight with 0.75.The total amount dumped by 53 trucks at the Rajbandh site during these 3 days is:

(14 x 6.228 + 25 x 7.747 + 9 x 4.926 + 2 x 3.84 + 1 x 0.766 + 2 x 0.575) = 326.86 tones.

As per KCC information, 36 trucks ply everyday carrying wastes i.e. the remaining (36 x 3 –53) = 55-truck load of wastes were dumped at the unofficial sites. Since the number of trucks for the Rajbandh site and the unofficial sites are 53 and 55 respectively, the waste dumped at the unofficial sites is assumed to be the same as that of Rajbandh i.e. 326.86 tones. The total amount of wastes carried by KCC trucks is 326.8 + 328.68 = 653.72 tones for 3 days i.e. 653.72 ÷ 3 = 217.91 tones per day. According to KCC officials and interview with some NGOs and other stakeholders, KCC support facilities are grossly insufficient to cope with the situation and as such 30 – 50% of the waste generated in the city is handled by KCC trucks. The waste generated per day in the city therefore lies between 463 tones and 726 tones, the average being 595 tones. The population of Khulna Statistical Metropolitan Area (KSMA) was 1.34 million in 2001 with a compound growth rate of 2.96% over the period 1991 – 2001.(Census, 2001). Assuming the same growth rate, the population in 2005 stands at 15.06 million and the per capita waste generation per day therefore stands at 0.40 kg which is less than that in Dhaka (0.52 kg) (BCAS, 1998). The variation of per capita generation from 0.22 kg to 0.75 kg quoted in the literature arose from area and season specific surveys; in the affluent areas and during the wet season the value will be high, while in the slums and during dry season it will be low.

Mismanagement Practices

Figure 1 Misplacement of solid wastes around Rajbandh dumping site

- Waste are dumping near the road site thus foul odor are emitted which pollute the environment.
- Blocking of drainage systems resulting in wastewater overflow during rainy season due to dumping the waste at un-official site.
- Pollution of surface water bodies and groundwater.
- Spreading of wastes by scavenging birds and animals.
- Indiscriminate disposal of hospital wastes that contain pathogenic organisms which may lead to spread of infectious diseases.
- Transmission of vector-borne diseases.
- Health risks to solid waste workers and scavengers.

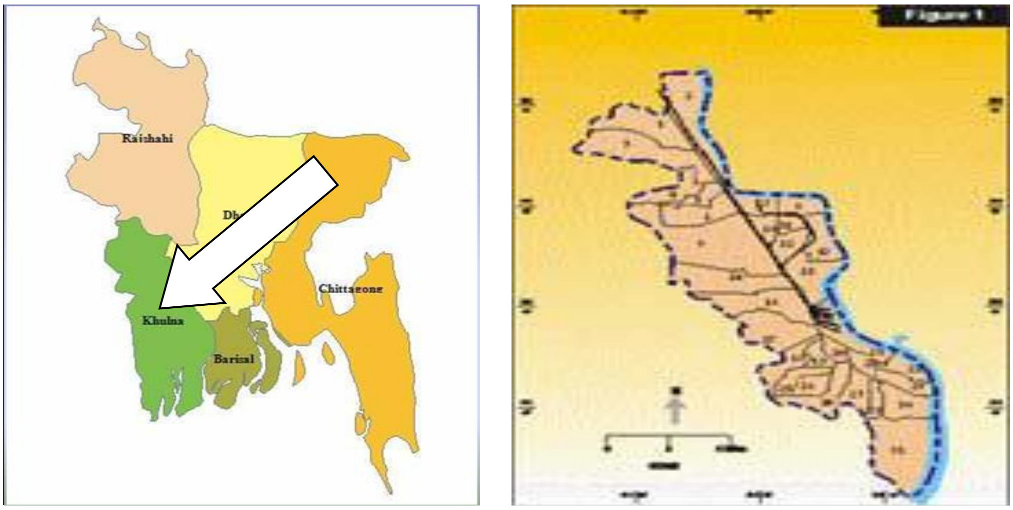
Due to poor transportation facilities rutting is formed during rainy season when heavy loaded truck is moving over the muddy road. The waste water is accumulated on the rutting and egress into the groundwater thus contamination is occurred.

Figure 2 L) Unplanned transportation facilities R) Burning of waste causes environmental pollution

This is the hospital waste burning chamber but in this burning chamber hospital as well as non bio degradable waste is also burned. So as a result various chemical compounds are emitted in the and thus air is contaminated. Specified temperature is not maintained in this burning chamber.

Khulna, the third largest metropolitan city of Bangladesh, stands on the banks of the Rupsha and the Bhairab rivers. It is in the south-western part of the country with its location on the axis of Jessore-Mongla port, the second largest seaport of the country.Geographically, Khulna lies between 22047´16´´to 22052´ north latitude and 89031´36´´ to 89034´35´´ east longitude. The city is 4 m above the mean sea level (MSL). At present, Khulna city has a population of about 1.5 million with an area of 47 square kilometers and 31 Wards.Khulna is also an important river port city of Bangladesh. Well-connected by the rivers Rupsha and Bhairab and located at the lower extreme of the Ganges delta, Khulna city acts as a place of trade and commerce and production centre of the region. Khulna obtained its status as a formal town after the establishment of the municipality in 1884 during the British colonial regime. During late 1950s and 1960s Khulna became an important centre for industrial development. Many industries such as newsprint mills, shipyard, jute mills, match factories, jute bailing presses, hardboard mills, etc. were established and associated commercial activities also increased manifold. Khulna attained the status of a City Corporation in 1990.

Over the period, Khulna city experienced continuous population growth accompanied by periodic changes to its territorial area. Again, due to the establishment of Khulna University, Khulna Medical College and Bangladesh Institute of Technology and Teacher’s Training College, Agricultural Training Institute, etc. along with increased activities resulting from the expanding shrimp export, Khulna has gained tremendous potential for further socio-economic activities and physical development. All these developments will have far reaching impact on the overall environment of Khulna city. Khulna city is located on natural levees of the Rupsha and Bhairab rivers and characterized by Ganges tidal floodplains with low relief, criss-crossed by rivers and water channels and surrounded by tidal marshals and swamps. The impact of urbanization in Khulna in terms of mass poverty, gross inequality, high unemployment, under-employment, and proliferation of slum areas and squatters and general deterioration in overall environmental conditions have become the major concerns of the policy issues. There is clear evidence that water is in short supply, there is unhygienic sanitation conditions and high incidence of diseases. Because of limitation of time and resources, it was impossible to survey the whole of the city. All these waste are dumped in the Rajbandh disposal site in order to further treatment and the residue is disposed in the sanitary landfill. For this reason, to understand the solid waste management and the tradition means of response of the people towards solid waste management at Rajbandh in KCC is selected



Figuure 3 Location of Rajbandh Waste Disposal Site

Rajbandh Trenching Ground Site

It is at present the only official dumping site, 25 acres in area situated at a distance of about 10 kms to the west of KCC Headquarter. Field investigators were stationed at the dumping sites to collect information on the nos. of trucks, capacity measurement of trucks, nature of solid waste and the origin of the waste. Data were collected on 16th, 17th and 18th June 2005. Four field investigators were engaged there from 6.00AM to 12:00 midnight for actual waste transportation. 16-19 waste carrying trucks were employed per day at Rajbandh Trenching Ground. Attempts were made to determine the weights of truck with load and without load to determine the actual load. Unfortunately, no scale is at present available in the Khulna city to carry out the measurement. Bulk/packing density was therefore, measured separately by making a container of 8 cft. volume (2ft.x 2ft .x 2ft.). As shown in Table 1 average bulk/packing density is found to be 15.5 kg/cft. i.e. 547.31 kg/m3.

Table 1 Bulk/ Packing density data of Khulna city solid waste and actual weights of MSW per truck

i	Measurement i (Net weight of MSW)=116 kg
ii	Measurement ii (Net weight of MSW)=124 kg
iii	Measurement iii (Net weight of MSW)=132 kg
	TOTAL= 372 Kg
	Average weight= 124 Kg
	Bulk/Packing density=15.5Kg/cu ft. or 547.31 Kg/m^3

Name plate capacity	7 tones	6 tones	5 tones	1.5 tones
Truck volume	11.38	9.00	6.00	1.40
Actual weight with full load (Kg)	6.228	4.926	3.283	0.766
Actual weight with 20% over loading (tones)	7.474	5.911	3.940	0.919
Actual weight with 75% loading (tones)	6.471	3.694	2.462	0.575

Table 2 Summary report of waste quantity dumped at Rajbandh, Khulna

Time window		7 tones capacity truck (Nos.)			6 tones capacity truck (Nos.)			5 tones capacity truck (Nos.)			1.5 tones capacity truck (Nos.)			Total truck	Total tones
Time	Observing time (hrs)	Full	Over	Half	Full	Over	Half	Full	Over	Half	Full	Over	Half		
6.0AM-12.0 AM	6	6	17	--	9	--	--	1		--	--	--	--	33	211.94
12.0 AM-18.0 PM	6	5	4	--		--	--	1	--	--	1	--	2	13	65.82
18.0 PM-24.0PM	6	3	4	--	--	--	--	--	--	--	--	--	--	7	48.55
Total trucks observed in	18	14	25					2	-	-	0.76	-	2	53	--
T. tones		87.1	186.7		44.3			6.56							326.31

(18-hrs. continuous data collection on each day of 16 June, 18 June 2005. Started at 6:00 AM and continued up to 12:00 midnight every day). There was no transportation of wastes between 24: 00 hrs and 6:00 hrs*)*

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Socio-Economic Condition of Rajbandh

Rajbandh is an important hub concerning the socio-economic potentialities. People of different categories and of different religion are living in this region. It is at present the only official dumping site, 25 acres in area situated at a distance of about 10 kms to the west of KCC Headquarter. The actual numbers of different institutional, commercial and other landmarks are summarized in Table 3 as under.

Table 3 General information on the infrastructures at Rajbandh in Khulna

Items	Nos.	Items	Nos.
Educational Institution	8	Children park	1
Clinic, hospital and pathological Laboratory	6	Graveyard	2
Govt. semi govt. & non govt. office	6	Market	3
NGO	13	Shopping complex	1
Newspaper office	1	Drain	89
Police box	2	Temple	1
Prayer centre	31	Mosque	2

METHODOLOGY

The field investigation for inspection of the management system of Municipal Solid Waste and finding the mismanagement at Rajbandh dumping site. Collecting the water sample from deep tubewell surrounding the disposal site for evaluating the groundwater quality that may spread any diseases among the people living around the dumping site. Field observation was done aiming to find out the mismanagement practice at Rajbandh dumping site. Finally recommend the sustainable management practice at disposal site and safe the area from various possible contaminations.Collecting the water sample from deep tubewell (90 ft deep) surrounds one km radius of dumping area. Taking this water sample into the laboratory for examining the water quality whether it is suitable or not. Then collecting the water sample from two and three km radius and finding the relationship between one, two and three km radius data whether it is increasing or decreasing.

DO bottle is required for laboratory analysis of BOD_5. DO meter is needed for measurement of BOD_5, it is measured by mg/l. For determination of COD $k_2cr_2o_7$ is taken in the pipette. Ferrion indicator is used as reagent and titrates until the radish color formed. p^H meter is used for determination of P^H. Conductivity meter is used for determination of conductivity. In burette H_2SO_4 (0.02N) is used. Methyl orange is used as a indicator and titrates until the pink color is formed. It is measured by mg/l. EDTA is used in burette for titration in Hardness test. Erichrom black T (EDT) is used as a reagent and titrates until the blue color is formed. For TDS test the sample is taken into oven at 105^0c. For cl^{-1} test potassium chromate is used as a indicator and titrates until radish color is formed. For Iron (Fe) test ferrover iron reagent is used and spectro-photometer 2500 instrument is used whose code No.265. For determination of Nitrate (NO_3) nitraver 5 nitrate reagent is used and instrument code NO.353. Phosver phosphate is used as a reagent for determination of PO_4 and the instrument code NO.490.

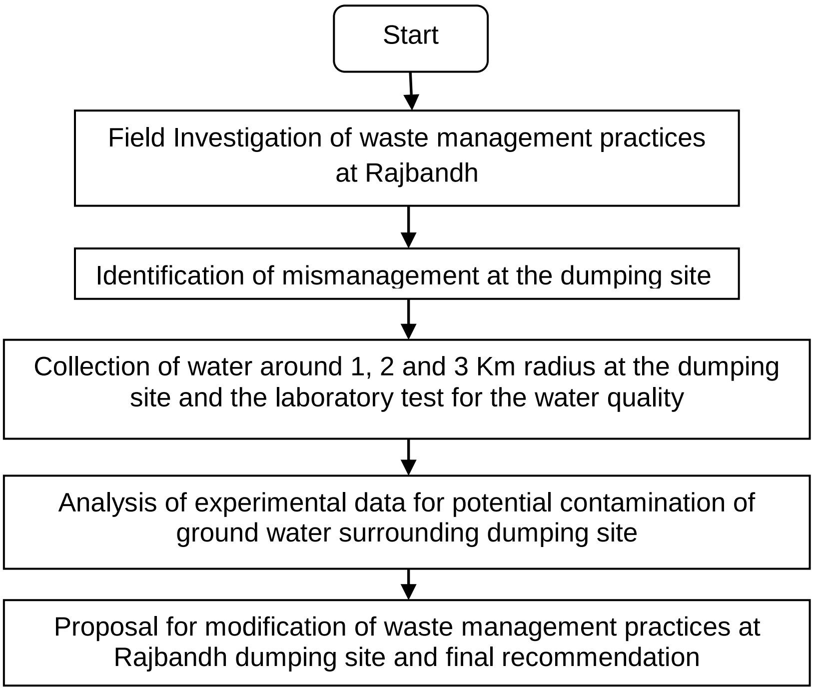


Figure 4 Flow chart of working plan

RESULT AND DISCUSSION

Preliminary field inspection at Rajbandh dumping site was done to notice the mismanagement practices and recommending the possible solution for ensuring the management practices. Organic matter greatly influences the mobility of leachate in aqueous environments through both chemical and biological processes. When the leachate ingress into the groundwater then it contaminates the groundwater. Different graphs are plotted for representing the groundwater contamination around the Rajbandh dumping site that are shown in Figure 5~8.

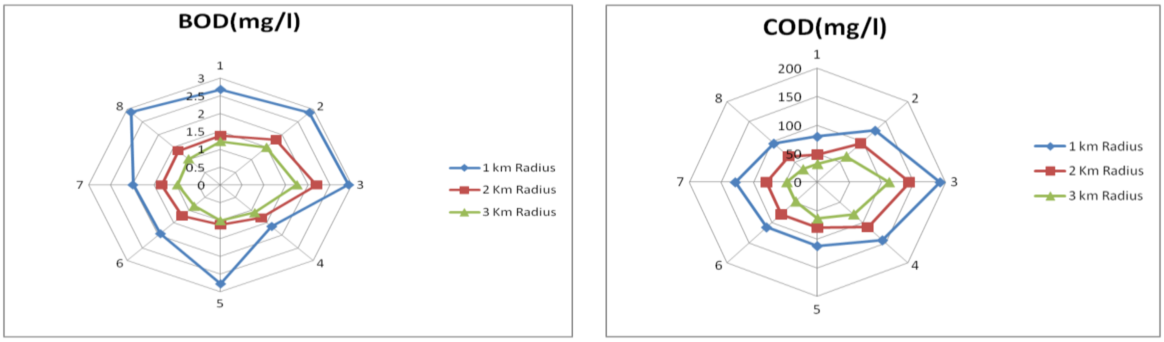


Figure 5 Graphical representations of BOD and COD around Rajbandh waste dumping site

The Figure 5 (L) shows that BOD value around 1 km is less then gradually it is rising around two & three km radius. In Bangladesh standard BOD_5 is 0.2 mg/l. Again the Figure 5 (R) shows that the COD value, is less in one km but it is rising gradually in two and three km radius. Large amount of COD in water indicate the low quality of groundwater. In Bangladesh standard COD_5 is 4 mg/l. So it is a clear indication that leachate contaminate the groundwater.

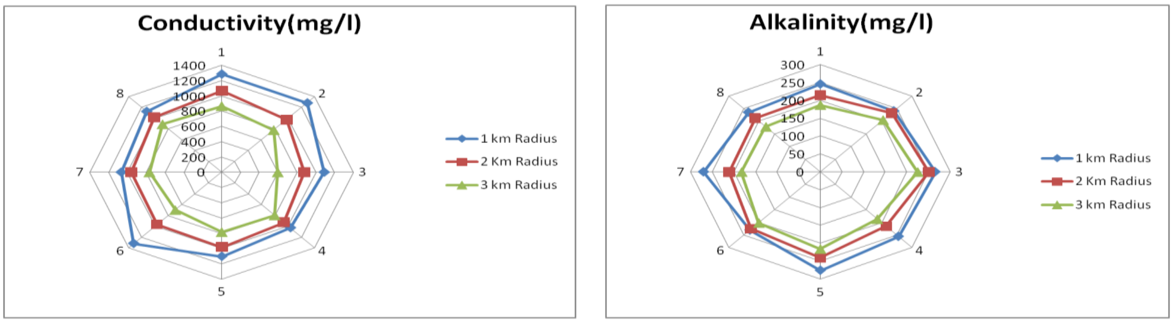


Figure 6 Graphical representation of Conductivity and alkalinity around Rajbandh waste dumping site

The Figure 6 (L) shows that conductivity value is increasing gradually one, two and three km radius. In one km radius conductivity less means the quality of groundwater is low because of having position near the waste dumping site. This value is gradually raising means groundwater is less affected by leachate because of having longer distance from waste dumping site. Figure 6 (R) shows that the Alkalinity value is less in one km but it is rising gradually in two and three km radius. So it is a clear indication that leachate contaminate the groundwater.

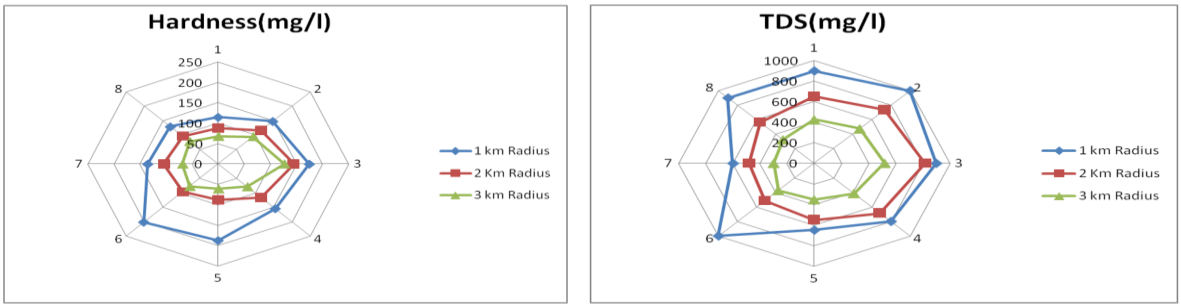


Figure 7 Graphical representations of Hardness and TDS around Rajbandh waste dumping site

Bangladesh standards hardness value is 200-500 mg/l as $CaCO_3$. The Figure 7 (L) shows that conductivity value is increasing gradually one, two and three km radius. In one km radius Hardness less means the quality of groundwater is low because of having position near the waste dumping site. This value is gradually raising means groundwater is less affected by leachate because of having longer distance from waste dumping site. The Figure 7 (R) shows that the TDS value is less in one km but it is rising gradually in two and three km radius. So it is a clear indication that leachate contaminate the groundwater.

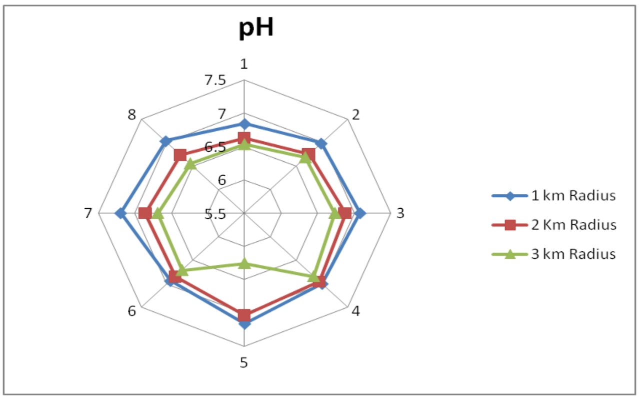


Figure 8 Graphical representations of pH around Rajbandh waste dumping site

The Figure 8 shows that the pH value is below the Bangladesh standard in one km but it is rising gradually in two and three km radius. So it is a clear indication that leachate contaminate the groundwater.

Summary of Groundwater Quality around Rajbandh Dumping Site

On the basis of experiment result in the laboratory of collected water around (1, 2 and 3 km) Rajbandh damping site. The following parameter values are found in the laboratory at different direction that is given in the Table 4~6.

Table 4 One km radius at Rajbandh dumping site

Location	BOD (mg/L)	COD (mg/L)	Conductivity (µs/cm)	Alkalinity (mg/L)	Hardness (mg/L)	TDS (mg/L)	P^H
N	2.68	80	1286	245	115	900	6.84
NE	2.87	128	1283	240	148.2	1000	6.98
E	2.92	192	1086	265	175	900	7.08
SE	1.65	144	1030	255	155	800	7
S	2.78	112	1104	275	188	650	7.16
SW	1.94	112	1321	230	201.9	1000	6.94
W	2	128	1066	270	135	600	7.2
NW	2.89	96	1126	235	129.6	900	7.03

Table 5 Two km radius at Rajbandh dumping site

Location	BOD (mg/L)	COD (mg/L)	Conductivity (µs/cm)	Alkalinity (mg/L)	Hardness (mg/L)	TDS (mg/L)	P^H
N	1.39	48	1068	215	88	650	6.63
NE	1.79	96	972	232.5	116.5	735	6.75
E	2.19	144	876	250	145	820	6.88
SE	1.31	112	929	215	116.5	685	6.95
S	1.12	80	982	240	88	550	7.03
SW	1.23	80	971	225	96.5	515	6.84
W	1.34	80	960	210	105	480	6.86
NW	1.36	64	1014	212.5	96.5	565	6.74

Table 6 Three km radius at Rajbandh dumping site

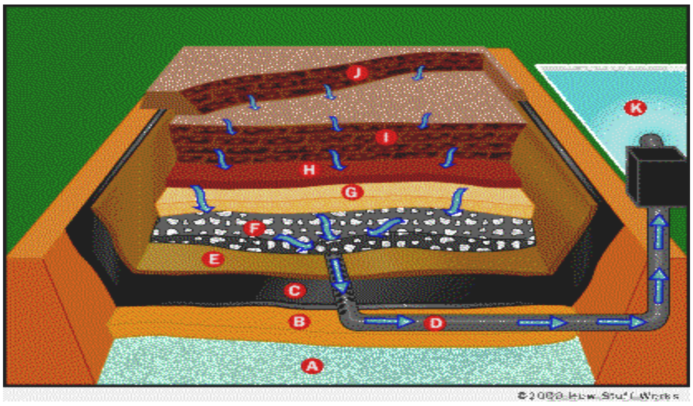
Location	BOD (mg/L)	COD (mg/L)	Conductivity (µs/cm)	Alkalinity (mg/L)	Hardness (mg/L)	TDS (mg/L)	P^H
N	1.22	32	867	187	68	425	6.53
NE	1.49	64	782	205	95	475	6.68
E	1.75	112	596	225	127	520	6.74
SE	1.11	80	796	186	79	415	6.84
S	1.01	64	782	215	61	355	6.25
SW	0.85	48	698	201	78	375	6.71
W	0.98	48	768	182	69	298	6.69
NW	1.03	32	889	178	78	325	6.55

CONCLUSION

- Preliminary field inspection at Rajbandh dumping site was done in order to notice the mismanagement practices.
- The water samples around Rajbandh dumping site shows that the value of water quality parameter are increasing gradually three, two and one km radius. Maximum biochemical oxygen demand (BOD) value around one, two and three km radius is 2.92, 2.19 and 1.75 mg/l respectively. In Bangladesh standard BOD_5 is 0.2 mg/l. The value of other water quality parameter such as Chemical Oxygen Demand (COD), Electrical Conductivity, Alkalinity, Hardness and Total Dissolved Solid (TDS), is increasing respectively three, two and one km radius. So it is a clear indication that leachate contaminate the groundwater.
- Constraints in the Existing Management System are misplacement of solid wastes around Rajbandh dumping site, unplanned transportation facilities, and burning of waste causes environmental pollution.
- Dumping of solid wastes in a particular place. Strict law is enforcing for ensuring the sustainable management practice at Rajbandh dumping site. Ensuring good transportation facilities by providing rigid or flexible pavement with appropriate roadway capacity instead of muddy road. Provide an appropriate burning chamber for hospital and other hazardous waste. Non degradable waste such as polythene is not burned in burning chamber thus avoiding the release of harmful chemical component. Sanitary landfill is provided for the protection of groundwater from leachate contamination of solid waste

RECOMMENDATION

Landfills may include internal waste disposal sites (where a producer of waste carries out their own waste disposal at the place of production) as well as sites used by many producers. Many landfills are also used for other waste management purposes, such as the temporary storage, consolidation and transfer, or processing of waste material (sorting, treatment, or recycling).



A: Ground Water,
B: Compacted Clay,
C: Plastic Liner,
D: Leachate Collection Pipe
E: Geotextile Mat,
F: Gravel,
G: Drainage Layer,
H:-Soil Layer,
I:-Old Cells,
J: New Cells &
K: Leachate Pond.

Figure 9 Cross-section of the structure of a municipal solid waste Landfill

Material shall be a woven polypropylene fabric with minimum thickness of 1.5 mm (0.06 inch), minimum width of 3.7 m (12 ft) and shall have minimum tensile strength of 0.67 kN (warp) 0.36 kN (fill) in conformance with the requirements in ASTM Designation: D 4632. The permittivity of the fabric shall be approximately 0.07 sec –1 in conformance with the requirements in ASTM Designation: D4491. The fabric shall have an ultraviolet (UV) stability of 70 percent in conformance with the requirements in ASTM designation: D4355. Geotextile blankets shall be secured in place with wire staples or sandbags and by keying into tops of slopes and edges to prevent infiltration of surface waters under Geotextile. Staples shall be made of 3.05-mm (0.12-inch) steel wire and shall be U-shaped with 200-mm (8-inch) legs and 50-mm (2-inch) crown.

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Analysis and Selection of the Appropriate Aggregation Function for Calculating of Leachate Pollution Index

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ABSTRACT

A technique to quantify the leachate pollution potential of landfills on a comparative scale using an index known as the Leachate Pollution Index (LPI) has been developed elsewhere. The LPI is a quantitative tool by which the leachate pollution data of the landfill sites can be reported uniformly. It is an increasing scale index and has been formulated based on the Delphi technique. The formulation process involved selecting variables, deriving weights for the selected pollutant variables, formulating their subindices curves and finally representing the pollutant variables to arrive at the LPI. The aggregation function is one of the most important steps in calculating any environmental index. If aggregation function is ambiguous, the result will raise an unnecessary alarm, indicating a comparatively less polluted environmental situation as mere contaminated. Similarly, if the aggregation function is eclipsed a false sense of security may be created, indicating a highly polluted environmental situation as less polluted. In this paper, the concept of LPI is described in brief and the various possible aggregation functions are described and used to calculate LPI values for an actual landfill site to select the most appropriate aggregation function. Based on the results, it is concluded that the weighted linear sum aggregation function is the best possible aggregation function for calculating LPI. Sensitivity analysis of the six short-listed aggregation functions is performed to substantiate this conclusion.

Keywords: *Landfill lysimeter, solid waste, leachate, aggregation function, leachate pollution index.*

INTRODUCTION

Leachate Pollution from closed and active landfills is an important issue, as it affects human health and the environment to a great extent. The leachate produced from a landfill may enter the underlying groundwater or the adjoining surface water bodies and can seriously degrade the water quality (Chian and DeWalle 1976; Lo 1996; Masters 1998). Groundwater, once contaminated is difficult if not impossible to improve. It has already become necessary to shut down thousands of drinking water wells across the United States due to the contamination from landfills. The problem is more acute in the underdeveloped and developing nations, where the landfills do not have any base liners or leachate collection and treatment systems. A strong need is presently being felt to take appropriate remedial measures to avoid contamination of the underlying soils and groundwater water aquifers from the leachate generated from the landfills. State regulatory authorities, in almost all the countries in the world, have framed regulations to safeguard against the contamination of groundwater sources from the leachate generated from the landfills. But necessary remedial and preventive measures can't be undertaken at all the existing closed and active landfill sites in one go because of financial constraints. In an effort to quantify the leachate pollution potential of the landfill sites, an index known as the Leachate Pollution Index (LPI) was developed using the Delphi technique (Kumar and Allapat 2003).

The formulation of an environmental index involves four basic steps: (1) Selection of variables; (2) Derivation of weights; (3) Formulation of their subindice equation and (4) Aggregation of the subindices. The aggregation process is one of the most important steps. It is here where most of the simplification (reduction of information) takes place and most of the distortion is likely to be introduced (Ott 1978). In this paper, the concept of LPI is described in brief and various possible aggregation methodsare reviewed and applied in an effort to select the most appropriate one for calculating LPI. The LPI values based on the leachate characteristics of a landfill site in India are calculated using the various aggregation functions. Sensitivity analysis of six aggregation functions is also performed to select the most appropriate aggregation function.

Table 1 Weights of pollutant variables included in leachate pollution index (Kumar and Alappat 2003)

Number	Pollutant	Pollutant weight
1	Chromium	0 .064
2	Lead	0.063
3	COD	0.062
4	Mercury	0.062
5	BOD_5	0.061
6	Arsenic	0.061
7	Cyanide	0.058
8	Phenolic compounds	0.057
9	Zinc	0.056
10	pH	0.055
11	TKN	0.053
12	Nickel	0.052
13	Total Coliform bacteria	0.052
14	Ammonia nitrogen	0.051
15	Total dissolved solids	0.050
16	Copper	0.050
17	Chlorides	0.049
18	Total iron	0.045
	Total	1.000

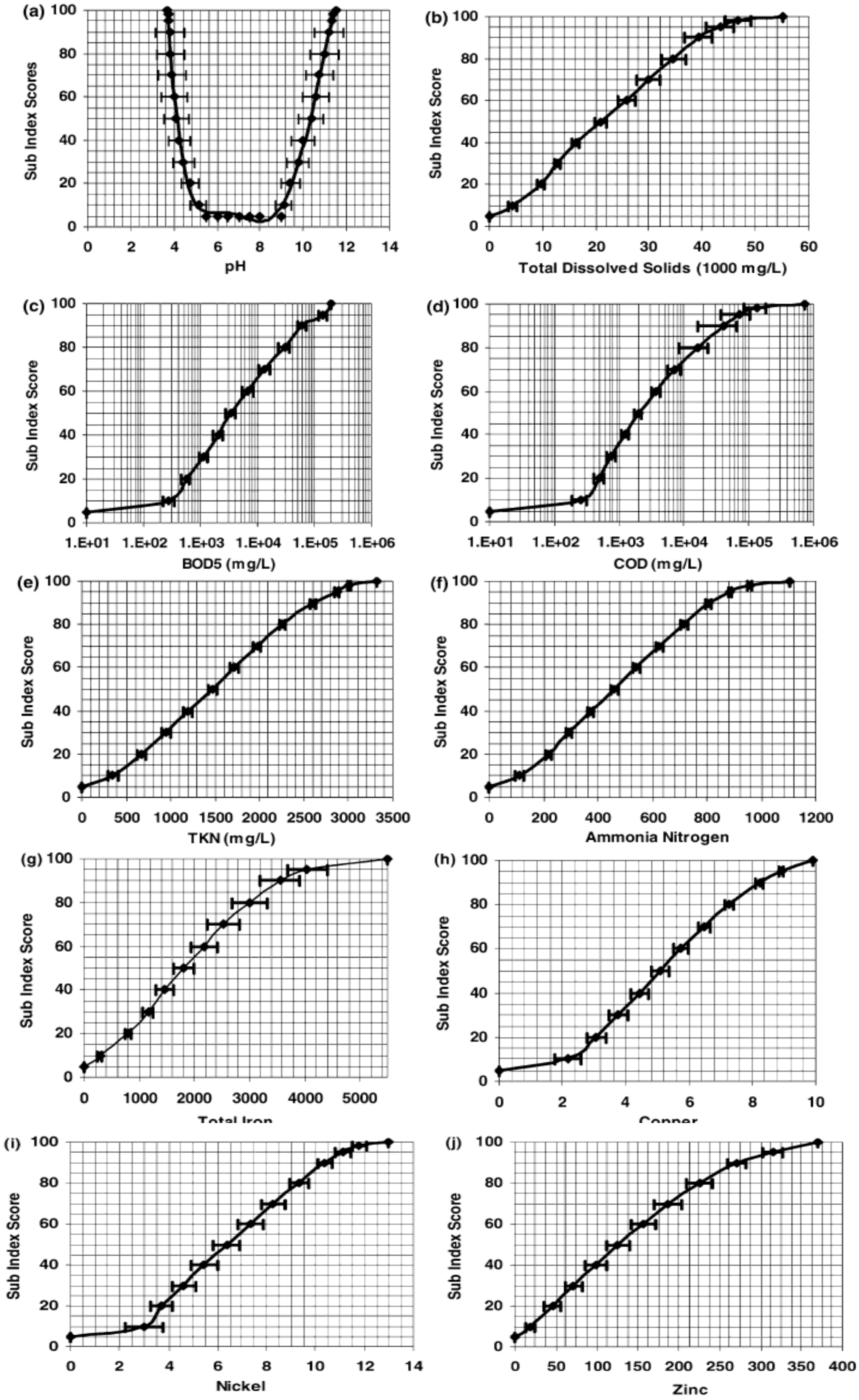
CONCEPT OF LEACHATE POLLUTION INDEX

In an effort to develop a system to compare the leachate contamination potential of various landfill sites in a given geographical area, 80 panelists, which included academicians in environmental engineering, environmental regulatory authority scientists, consulting engineers, and members of the International Solid Waste Asociation (ISWA) from around the world, were surveyed (Kumar and Alappat 2003). The survey was conducted using multiple questionnaires to develop a LPI.

- **i.** The index is a mathematical method of calculating a single value from multiple chemical and boilogical test results of the landfill leachate.
- **ii.** The single value LPI have a grade that expresses the overall leachate contamination potential of a landfill, based on several leachate pollution parameters at a given time.
- **iii.** It is an increasing scale index, wherein a higher index value indices a poorer environmental condition.

The 18 leachate pollution parameters selected for inclusion in the LPI, based on the survey of panelists, were chromium, lead, chemical oxygen demand (COD), mercury, 5 day biochemical oxygen demand (BOD_5), arsenic, cyanide, phenolic compounds, zinc, pH, total Kjeldahl nitrogen, nickel, total Coliform bacteria, ammonical nitrogen, total dissolved solids (TDS), copper, chlorides, and total iron((Kumar and Alappat 2003). The weights for these parameters were calculated based on the significance levels given by the panelists for these parameters on a scale of 1 to 5 and are shown in Table 1.

A selected group of panelists were asked to draw curves for the pollutant variables included in the LPI with respect to leachate pollution ranging from 5 (best) to 100 (worst). Levels of leachate pollution from 0 to 100 were indicated on the ordinate of each graph, while various levels of concentration of the particular variable, up to the maximum limits reported in literature, were indicated on the abscissa. The curves drawn by the panelists were averaged to obtain "average subindex" curves for each parameter. The averaged subindex curves are shown in Fig. 1.



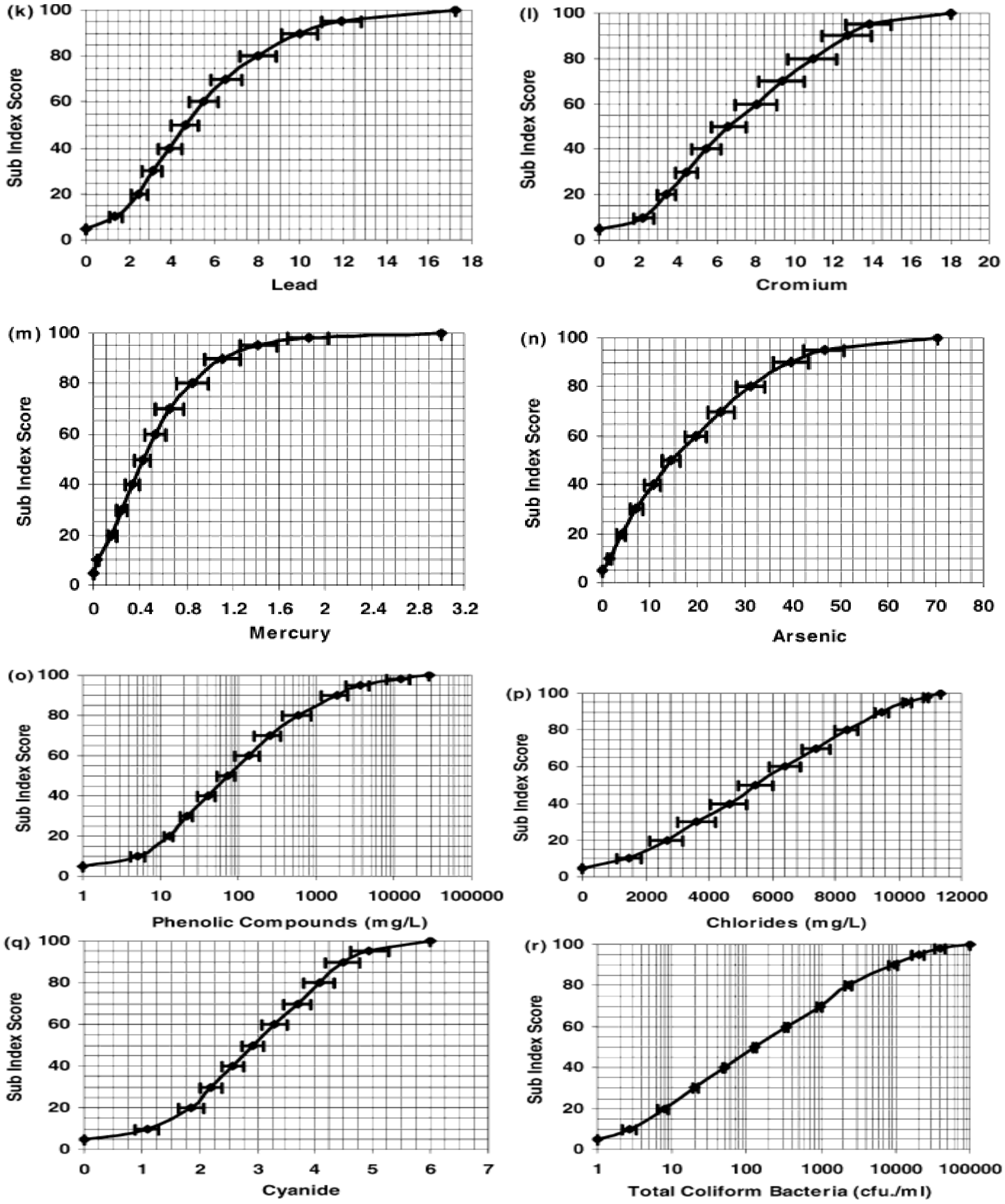


Figure 1 The averaged sub index curves of pollutant (a) pH (b) TDS (c) BOD_5 (d) COD (e) TKN (f) ammonia nitrogen (g) iron (h) copper (i) nickel (j) zinc (k) lead (l) chromium (m) mercury (n) arsenic (o) phenol (p) chlorides (q) cyanide (r) TCB*(after Kumar and Alappat 2003)*.

AGGREGATION FUNCTION

Aggregation methods are crucial in the field of environmental indices, as they affect the quality of result in many ways .Aggregation has been defined as “the process of adding variables or units with similar properties to come up with a single number that represents the approximate overall value of its individual component.”

Aggregation function s usually consists of either of the following three forms:

I. Additive form (summation function), in which individual variables are added together;

II. Multiplicative form (multiplication function) in which a product is formed of some or all of the variables and

III. Maximum or minimum operator form, in which just the maximum or the minimum subindex value of the variable directly accepted.

The type of aggregation function is selected based on the function and the use of the index. Because most of the air pollution indices reported in the literature are of the increasing scale form, they mostly use the additive form aggregation function (Babcock and Nagda 1972; Inhaber 1974; Swamee and Tyagi 1999) or the maximum operator form aggregation function (Ott 1978). Some of the water quality indices are of the decreasing scale from (Horton 1965; Brown et al. 1970; Dec et al. 1973; Walski and Parker 1974; Smith 1990), and the others are of the increasing scale form (Prati and Pesarin 1971). The water quality indices, independent of their functional forms, use all three forms of aggregation functions. The additive form of aggregation function was used by Brown et al. (1970) for developing a water quality index for the U.S. National Sanitation Foundation; by Horton (1965) for Horton's Water Quality Index; by Prati and Pesarin (1971) for Prati'sI, plicit Index of Pollution; by Prati and Pesarin (1971) for Prati's Implicit Index of Pollution; by Truett et al. (1975) for MITRE's National Planning Priority Index; and by Gilianovic (1999) for the Water Quality Index for Dalmatia. The multiplicative form of aggregation function was used by Landwehr for the National Sanitation Foundation's Water Quality Index [Landwehr (1974), cited in Ott (1978)] and by Walski and Parker (1974) for Walski and Parker's Index. The minimum operator form of aggregation function was used by Smith (1990) for developing a decreasing scale water quality index for New Zealand. Harkins (1974) proposed an aggregation function based on Kendal's nonparametric multivariate ranking procedure for the National Sanitation Foundation water quality index developed by Brown et al. (1970), which was later criticized by Landwehr and Deininger (1976).

PROCEDURE FOR SELECTING APPROPRIATE AGGREGATION FUNCTION

The following aspects are to be considered for selection of the appropriate aggregation method.

Functional Form of Index

An index can be an increasing scale index or a decreasing scale index. In the case of an increasing scale index, usually called an "environmental pollution index,"higher values indicate a worse state than lower values. In the decreasing scale indices, higher values are associated with a better state than lower values and are usually referred to as "environmental quality indices."

Strength and Weakness of Aggregation Function

The two potential problems associated with aggregation functions are (Ott 1978):

1. An overstimation (ambiguity) problem, where the aggregate index I exceeds the critical level without any of the subindices exceeding the critical levels.

2. An underestimation (eclipsing) problem, where the aggregate index I does not exceed the critical level despite one or more of the subindices exceeding the critical levels.

These two problems crop up only with dichotomous subindices. The most appropriate aggregation function will minimize one or more both the overestimation and underestimation problems.

Parsimony Principle

when competing aggregation functions produce similar results with respect to overestimation and underestimation , the most appropriate aggregation function will be that which is mathematically simple (Joilands et al. 2003).

Transparency of Aggregation Function

Finally, an aggregation approach is successful if all assumptions and sources of data are identified, the methodology is transparent and publicly, and an index can be readily disaggregated into the separate components with no information lost (Hammond and Adriaanse 1965). In addition to the aforementioned procedure, the aggregation function selected for any environmental index shall also meet the following criteria. It should:

1. Be sensitive to the changes in an individual variable throughout its range;
2. Not be biased towards good or poor environmental quality ;
3. Consider weighting factors, as all variables included in the index are not equal contributors to environment pollution; and
4. Be relatively easy to use.

Selecting Appropriate Aggregation Function for Leachate Pollution Index

To select the most appropriate aggregation for LPI, the various possible aggregation functions are applied to landfill lysimeter leachate characterics at KUET campus. For the present study, the leachate characteristics of lysimeter at KUET campus , have been considered. The LPI values for the treated leachate have also been calculated using all the aggregation functions to demonstrate the behavior of various aggregation functions. The concentrations of the various leachate pollutant variables are shown in Table 3, Column 3. The different aggregation functions used are discussed and presented in Table 2.

Table 2 Aggregation functions used by different researchers for water quality and pollution indices

Equ. No.	Aggregation function	Function expression	Users	Specific remarks	Reported Table/column
1	Unweighted Additive Form	$LPI_{ua} = \frac{1}{n}\sum_{i=1}^{n} p_i$	Brown et al. (1970); Prati et al. (1971); McDuffie and Haney (1973)	Ambiguous function; shows eclipsing region; simple but little flexibility; unsuitable for dichotomous subindices.	Table 3, Column 5.
2	Weighted Linear Additive Form	$LPI_{wa} = \sum_{i=1}^{n} w_i p_i$	Horton (1965); Brown et al. (1970); Prati et al. (1971); Dinius(1972); Dee et al. (1973); Inhaber(1974); Ott (1978); Ball and Church (1980); Egborge and Coker (1986); Mohan et al. (1996); Giljanovic (1999); Prasad and Bose (2001); Bardalo et al. (2001); Kumar and Alappat(2004)	Ambiguity free function; shows small eclipsing with large number of variables; not suitable for dichotomous subindices; widely used aggregation function.	Table 3, Column 6.
3	Root Sum Power Additive form	$LPI_r = \left(\sum_{i=1}^{n} p_i^{\,r}\right)^{1/r}$	Swamee and Tyagi(1999); Kumar and Alappat(2004)	Shows reduced eclipsing but exhibit ambiguity problem; with increase in *r*, ambiguity decreases. If *r*→∞, it becomes ambiguity and eclipsity free function; use of aggregation function for*r*>2 is not practiced foraggregation of water pollution	

				indices.	
3(a)	Root Sum Power Additive form (r=2)	$LPI_2 = \left(\sum_{i=1}^{n} p_i^{\,2}\right)^{1/2}$			Table 3, Column 7.
3(b)	Root Sum Power Additive form (r=4)	$LPI_4 = \left(\sum_{i=1}^{n} p_i^{\,4}\right)^{1/4}$			Table 3, Column 8.
3(c)	Root Sum Power Additive form(r=10)	$LPI_{10} = \left(\sum_{i=1}^{n} p_i^{\,10}\right)^{1/10}$			Table 3, Column 9.
4	Weighted Root Sum Power Additive Form	$LPI_{wr} = \left(\sum_{i=1}^{n} w_i p_i^{\,r}\right)^{1/r}$	Kumar and Alappat (2004)	Exhibits slightly reduced ambiguity, unwidely used aggregation function.	
4(a)	Weighted Root Sum Power Additive Form (r=4)	$LPI_{w4} = \left(\sum_{i=1}^{n} w_i p_i^{\,4}\right)^{1/4}$			Table 3, Column 10.
4(b)	Weighted Root Sum Power Additive Form (r=10)	$LPI_{w10} = \left(\sum_{i=1}^{n} w_i p_i^{\,10}\right)^{1/10}$			Table 3, Column 11.
5	Root Mean Square Additive Form	$LPI_{rm} = \left(\sum_{i=1}^{n} \frac{1}{n} p_i^{\,2}\right)^{1/2}$	Inhaber (1974); Kumar and Alappat (2004)	Exhibits small ambiguity problems.	Table 3, Column 12.
6	Weighted Root Sum Square Aggregation Function	$LPI_{wr} = \frac{\left(\sum_{i=1}^{n} w_i p_i^{\,2}\right)^{0.5}}{\sum_{i=1}^{n} w_i}$	Inhaber (1975); Kumar and Alappat (2004)	Exhibits small eclipsing problems.	Table 3, Column 13.
7	Maximum Operator Function	$LPI_{max} = max.\{p_1, p_2, \ldots p_n\}$	Smith 91990); Swamee and Tyagi (2000); Kumar and Alappat (2004)	No eclipsing problem but exhibit ambiguity for large number of variables; suitable for aggregation of air pollutionsubindices; limited application for water quality indices.	Table 3, Column 14.
8	Ambiguity And Eclipsity Free Aggregation Function	$LPI_{aef} = \sum_{i=1}^{n} \left(p_i^{\,1/r}\right)^{r}$	Swamee and Tyagi (1999); Kumar and Alappat (2004)	Eclipsity and ambiguity free function, limited application for air pollution indices; minimal ambiguity for *r*=0.4.	
8(a)	$\mathrm{LPI}_{2.5}$Ambiguity and Eclipsity Free Aggregation Function	$\mathrm{LPI}_{2.5} = \sum_{i=1}^{n} \left(p_i^{\,2.5}\right)^{0.4}$			Table 3, Column 15.
9	Weighted Ambiguity and Eclipsity Free Aggregation Function	$LPI_{w2.5} = \left(\sum_{i=1}^{n} w_i p_i^{\,2.5}\right)^{0.4}$	Kumar and Alappat (2004)	Eclipsity&Embiguity free function; limited application for leachate pollution indices.	Table 3, Column 16.
Equ. No.	Aggregation function	Function expression	Users	Specific remarks	Reported Table/column

10	Multiplicative Aggregation Function	$LPI_{wm} = \prod_{i=1}^{n} p_i^{w_i}$	Walski and Parker (1974); Ball and Church (1980); Bhargava(1983, 1985); Dinius(1987); Swamee and Tyagi(2000); Kumar and Alappat (2004)	Nonlinear; ambiguity free but exhibits eclipsing at low weights and increasing scale indices; insensitive when applied to large number of variables.	Table 3, Column 17.
11	Unweighted Multiplicative Aggregation Function	$LPI_{um} = \left(\prod_{i=1}^{n} p_i\right)^{1/n}$	Landwehr and Deininger (1976); Bhargava(1985); Kumar and Alappat (2004)	Exhibits small eclipsity problem, applied for comparison purposes only.	Table 3, Column 18.
12	Geometric Aggregation Function	$LPI_{gm} = \left(\prod_{i=1}^{n} p_i^{g_i}\right)^{1/\gamma}$	Walski and Parker (1974)	Nonlinear; ambiguity free but exhibits eclipsing at low weights and increasing scale indices; insensitive when applied to large number of variables.	

Table 3 LPI values for leachate characteristics of landfill lysimeter leachate at KUET campus, using different aggregation functions

Pollutant	w_i	Conc.	p_i	LPI_{ua}	LPI_{wa}	LPI_2	LPI_4	LPI_{10}	LPI_{w4}	LPI_{w10}	LPI_{rm}	LPI_{wr}	LPI_{max}	$LPI_{2.5}$	LPIw2.5	LPIwm	LPI_{um}
Cr	0.064	1.75	6.5	6.5	0.416	42.25	1.79E+03	1.35E+08	1.14E+02	8.62E+06	42.25	2.704	6.5	1.08E+02	6.89E+00	1.13	6.5
Pb	0.063	0.68	9	9	0.567	81	6.56E+03	3.49E+09	4.13E+02	2.20E+08	81	5.103	9	2.43E+02	1.53E+01	1.15	9
COD	0.062	8425	70	70	4.34	4900	2.40E+07	2.82E+18	1.49E+06	1.75E+17	4900	303.8	70	4.10E+04	2.54E+03	1.30	70
Hg	0.062	0.4	59	59	3.658	3481	1.21E+07	5.11E+17	7.51E+05	3.17E+16	3481	215.822	59	2.67E+04	1.66E+03	1.29	59
BOD_5	0.061	1398	35	35	2.135	1225	1.50E+06	2.76E+15	9.15E+04	1.68E+14	1225	74.725	35	7.25E+03	4.42E+02	1.24	35
As	0.061	0.01	5.5	5.5	0.3355	30.25	9.15E+02	2.53E+07	5.58E+01	1.55E+06	30.25	1.84525	5.5	7.09E+01	4.33E+00	1.11	5.5
CN	0.058	1.3	10	10	0.58	100	1.00E+04	1.00E+10	5.80E+02	5.80E+08	100	5.8	10	3.16E+02	1.83E+01	1.14	10
Phenol	0.057	4	8.5	8.5	0.4845	72.25	5.22E+03	1.97E+09	2.98E+02	1.12E+08	72.25	4.11825	8.5	2.11E+02	1.20E+01	1.13	8.5
Zn	0.056	1.3	5	5	0.28	25	6.25E+02	9.77E+06	3.50E+01	5.47E+05	25	1.4	5	5.59E+01	3.13E+00	1.09	5
pH	0.055	7.3	6	6	0.33	36	1.30E+03	6.05E+07	7.13E+01	3.33E+06	36	1.98	6	8.82E+01	4.85E+00	1.10	6
TKN	0.053	3000	98	98	5.194	9604	9.22E+07	8.17E+19	4.89E+06	4.33E+18	9604	509.012	98	9.51E+04	5.04E+03	1.28	98
Ni	0.052	0.23	8	8	0.416	64	4.10E+03	1.07E+09	2.13E+02	5.58E+07	64	3.328	8	1.81E+02	9.41E+00	1.11	8
TCB	0.052	8000	92	92	4.784	8464	7.16E+07	4.34E+19	3.73E+06	2.26E+18	8464	440.128	92	8.12E+04	4.22E+03	1.27	92
NH3	0.051	1300	100	100	5.1	10000	1.00E+08	1.00E+20	5.10E+06	5.10E+18	10000	510	100	1.00E+05	5.10E+03	1.26	100
TDS	0.05	12540	28	28	1.4	784	6.15E+05	2.96E+14	3.07E+04	1.48E+13	784	39.2	28	4.15E+03	2.07E+02	1.18	28
Cu	0.05	0.98	7	7	0.35	49	2.40E+03	2.82E+08	1.20E+02	1.41E+07	49	2.45	7	1.30E+02	6.48E+00	1.10	7
C^{l-}	0.049	3597	30	30	1.47	900	8.10E+05	5.90E+14	3.97E+04	2.89E+13	900	44.1	30	4.93E+03	2.42E+02	1.18	30
Fe	0.045	82	9.5	9.5	0.4275	90.25	8.15E+03	5.99E+09	3.67E+02	2.69E+08	90.25	4.06125	9.5	2.78E+02	1.25E+01	1.11	9.5
Total	1.001			587	32.2675	39948	3.03E+08	2.28E+20	1.61E+07	1.19E+19	39948	2169.577	100	3.62E+05	1.95E+04	18.04	4.79E+22
LPI value				32.61	32.27	199.87	131.93	108.61	63.36	80.82	47.11	46.53	100.00	167.30	52.05	18.04	18.20

Note*: TCB= total Coliform bacteria; TKN= total Kjeldhal nitrogen; and TDS total dissolved solids. All the values are in mg/L except for pH and TCB.*

Table 4 Leachate Pollution Index (LPI) values for treated leachate using different aggregation functions

Pollutant	w_i	Conc^b.	p_i	LPI$_{ua}$	LPI$_{wa}$	LPI$_2$	LPI$_4$	LPI$_{10}$	LPI$_{w4}$	LPI$_{w10}$	LPI$_{rm}$	LPI$_{wr}$	LPI$_{max}$	LPI$_{2.5}$	LPI$_{w2.5}$	LPI$_{wm}$	LPI$_{um}$
Cr	0.06	2	9	9	0.58	81	6.56E+03	3.49E+09	4.20E+02	2.23E+08	81	5.18	9	243.00	15.55	1.15	9
Pb	0.06	0.1	5	5	0.32	25	6.25E+02	9.77E+06	3.94E+01	6.15E+05	25	1.58	5	55.90	3.52	1.11	5
COD	0.06	250	10	10	0.62	100	1.00E+04	1.00E+10	6.20E+02	6.20E+08	100	6.20	10	316.23	19.61	1.15	10
Hg	0.06	0.01	6	6	0.37	36	1.30E+03	6.05E+07	8.04E+01	3.75E+06	36	2.23	6	88.18	5.47	1.12	6
BOD_5	0.06	30	6	6	0.37	36	1.30E+03	6.05E+07	7.91E+01	3.69E+06	36	2.20	6	88.18	5.38	1.12	6
As	0.06	0.2	5	5	0.31	25	6.25E+02	9.77E+06	3.81E+01	5.96E+05	25	1.53	5	55.90	3.41	1.10	5
CN	0.06	0.2	6	6	0.35	36	1.30E+03	6.05E+07	7.52E+01	3.51E+06	36	2.09	6	88.18	5.11	1.11	6
Phenol	0.06	1	5	5	0.29	25	6.25E+02	9.77E+06	3.56E+01	5.57E+05	25	1.43	5	55.90	3.19	1.10	5
Zinc	0.06	5	6	6	0.34	36	1.30E+03	6.05E+07	7.26E+01	3.39E+06	36	2.02	6	88.18	4.94	1.11	6
pH	0.06	5.5-9	5	5	0.28	25	6.25E+02	9.77E+06	3.44E+01	5.37E+05	25	1.38	5	55.90	3.07	1.09	5
TKN	0.05	100	6	6	0.32	36	1.30E+03	6.05E+07	6.87E+01	3.20E+06	36	1.91	6	88.18	4.67	1.10	6
Ni	0.05	3	10	10	0.52	100	1.00E+04	1.00E+10	5.20E+02	5.20E+08	100	5.20	10	316.23	16.44	1.13	10
TCB	0.05	3^a	10	10	0.52	100	1.00E+04	1.00E+10	5.20E+02	5.20E+08	100	5.20	10	316.23	16.44	1.13	10
NH3	0.05	50	7	7	0.36	49	2.40E+03	2.82E+08	1.22E+02	1.44E+07	49	2.50	7	129.64	6.61	1.10	7
TDS	0.05	2100	7	7	0.35	49	2.40E+03	2.82E+08	1.20E+02	1.41E+07	49	2.45	7	129.64	6.48	1.10	7
Cu	0.05	3	18	18	0.90	324	1.05E+05	3.57E+12	5.25E+03	1.79E+11	324	16.20	18	1374.62	68.73	1.16	18
Cl^-	0.05	1000	8	8	0.39	64	4.10E+03	1.07E+09	2.01E+02	5.26E+07	64	3.14	8	181.02	8.87	1.11	8
Fe	0.05	100^a	7	7	0.32	49	2.40E+03	2.82E+08	1.08E+02	1.27E+07	49	2.21	7	129.64	5.83	1.09	7
Total	1.00			136	7.47	1196	1.62E+05	3.61E+12	8.40E+03	1.81E+11	1196	64.61	18	3800.76	203.34	7.05	2.16E+15
LPI value				7.56	7.47	34.58	2.01E+01	1.80E+01	9.57E+00	1.34E+01	8.15	8.03	18	27.04	8.38	7.05	7.11

The leachate generated from the okhla landfill is neither collected nor treated at present. However, guidelines and standards do exist in India for the discharge of leachate from municipal landfills. The LPI values for the treated leachate are also calculated using all these aggregation functions. In fact, the data set considered is not for the treated leachate of any municipal landfill site, but it is for the maximum permissible discharge limits for the various pollutant variables according to Indian regulations ("Municipal" 2000), assuming these to be the characteristics of the treated leachate. The LPI values for the treated leachate using all the aggregation functions are computed and shown in Table 4.

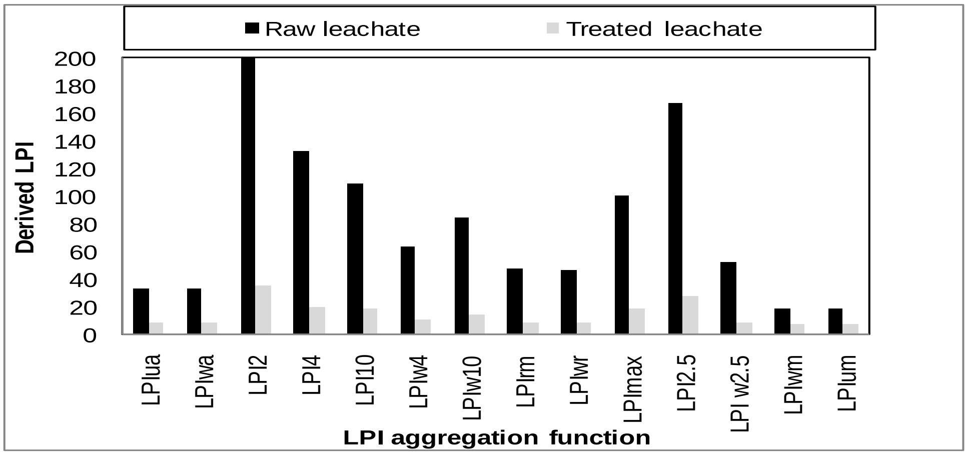


Figure 2 LPI values of raw and treated leachate of active landfill site in new Delhi, India, using different aggregation methods where series 1 indicates Raw leachate and series 2 indicates Treated Leachate.

RESULTS AND DISCUSSIONS

The LPI values for the landfill lysimeter at KUET campus, calculated using all the aggregation methods explained earlier are plotted in Figure 2. The LPI values of the treated leachate are also plotted in Figure 2.The LPI values calculated using various aggregation methods indicate that root sum power, 4th root power, 10th root power and the calculated LPI values exceed the maximum reported individual pollutant subindex value. All these values also exceed the theoretical range of LPI, i.e., 5-100. However, the 10th root power aggregation function results in the least ambiguous results, followed by the 4th root, the ambiguity and eclipsity free (2.5th root), and the root sum square aggregation functions. The maximum operator aggregation function does not show ambiguity of results, but it cannot be used for calculating the LPI values, as its results cannot be used to compare the fine gradations of leachate pollution. The results of the two multiplicative aggregation functions, the weighted multiplicative aggregation function and the unweighted multiplicative aggregation function, indicate high eclipsing of the data. The values are very low as compared to the other additive form aggregation functions. The unweightedadditive form and root mean square additive form aggregation functions suffer from the drawback that theweight of the variables are not considered, and all the variables are assumed to be of the same weight. Though the weighted linear sum aggregation function also suffers from the eclipsity problem, the eclipsity produced is small, as the number of the variables included in the aggregation function is large. The eclipsity problem is associated with this aggregation function when the dichotomous state of the index is to be reported, which is not the case here. The weighted sum aggregation, weighted root mean square, weighted 2.5th root, weighted 4th root, weighted 10th root, and weighted multiplicative aggregation functions, however, fulfill other criteria, such as weights of all the pollutant the variables being considered, and are easy to ascertain. The sensitivity of these six aggregation function to the changes of the individual pollutants is further conducted to select the best possible aggregation function.

Sensitivity analysis

The sensitivity analysis of the six aggregation functions, which takes into consideration the weights attached to the pollutants, with respect to the change in strength or concentration of two pollutants is performed independently. The six aggregation functions selected for the sensitivity analysis are-

1. Weighted sum additive aggregation function, LPIwa
2. Wighted root sum square aggregation function, LPIwr
3. Weighted 2.5throot (ambiguity and eclipsity free) aggregation function, LPIw2.5
4. Weighted 4th root sum aggregation function, LPIw4
5. Weighted 10th root sum aggregation function, LPIw10
6. Weighted multiplicative aggregation function, LPIwm

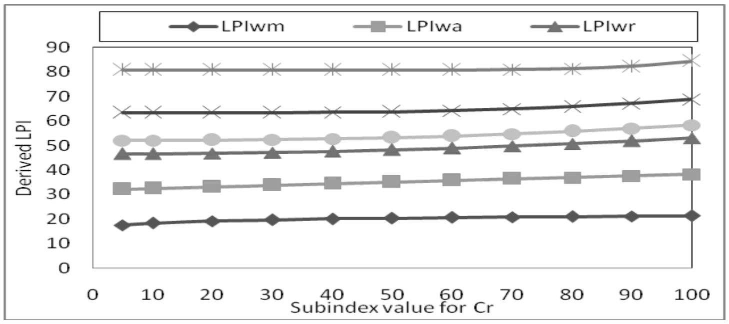


Figure 3 Sensitivity of weighted additive, weighted square root, weighted ambiguity eclipsity free, weighted fourth root, weighted tenth root, and weighted multiplicative aggregation function with respect to changes in subindex (concentration) values of chromium.

The two pollutants selected are chromium, which is the most significant variable and thus has the highest “weight” value, and total iron which is least significant pollutant with the lowest “weight” value. It will be hypothetical to assume that all the leachate pollutant variables included in LPI for a particular landfill will take on the highest possible subindex value of 100. Therefore, it is felt more practical to study the sensitivity analysis on actual landfill leachate data. For performing the sensitivity analysis, the subindex value of chromium is varied from 5 to 100 in the same data set of landfill lysimeter at KUET campus, and the LPI values using these six aggregation functions are calculated. The variations in the LPI values of this six aggregation functions with respect to the change in the subindex value of chromium are shown in Figure 3. The variations of the LPI values for all six aggregation functions with respect to the change in the subindex value of total iron from 5 to 100 for the same data set are shown in Figure 4.the calculated percentage variation of the LPI values over the minimum value for the subindex variation of chromium and total iron are shown in Table 5.

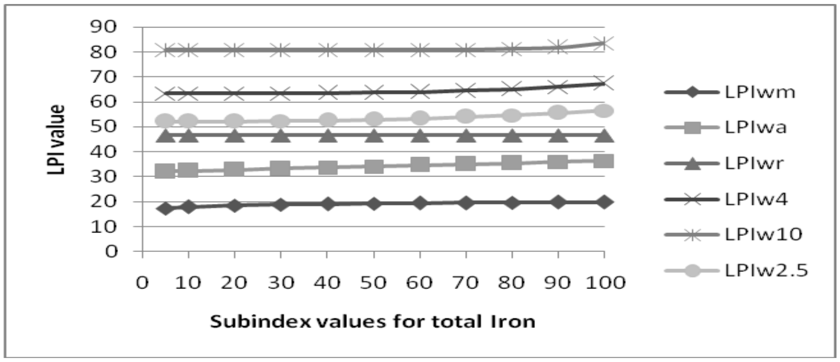


Figure 4. Sensitivity of weighted additive, weighted square root, weighted ambiguity eclipsity free, weighted fourth root, weighted tenth root, and weighted multiplicative aggregation function with respect to changes in subindex (concentration) values of total Iron.

From Table 5, it can be concluded that the weighted multiplicative aggregation function is the most sensitive one in comparison to other aggregation functions, showing change in LPI values of 21.18 and 14.48% for chromium and total iron, respectively. The next most sensitive aggregation function is the weighted linear sum aggregation function, which shows 18.89 and 13.31% variation in LPI values for the two pollutants, followed by the weighted root square,weighted 4th root, and weighted 10th root aggregation function. The weighted 10th root aggregation function is least sensitive to the subindex changes of chromium and total iron.

Further, the behavior of LPI_{w10}, LPI_{w4}, $LPI_{w2.5}$, and LPI_{wr} values with respect to the changes in subindex values of chromium and total iron, shown in Figures 3 and 4, clearly indicates that these aggregation function areleast sensitive to changes in the subindex values, particularly for the lower values of the subindex. The LPIw10value remains almost constant for a change in the subindex value of chromium from 5 to 70, while the LPIw4 value remains almost constant for a change in the subindex value of chromium from 5 to 60. Similarly, the LPIw2.5 and LPIwr are also insensitive for the changes in the subindex value of chromium from 5 to 40.The behavior of these aggregation functions with respect to the changes in the subindex values of total iron is similar. Therefore, these aggregation functions may not be useful when the fine gradation in leachate pollution of different landfill sites spread over a given area are to be compared.

Though the sensitivity analysis shows that the variation of LPIwm values for the change in subindex values of chromium and total iron is highest, it suffers from the drawback that the function is nonlinear. Figs. 3 and 4 indicate that the LPIwwm values hardly change when the subindex values of chromium and total iron vary from 50to 100, but the changes in LPIwm values is more rapid when the subindex value changes from 5 to 50. Thus the LPIwm curve does not represent the change in subindex values as effectively as LPIwa. Moreover, the weighted multiplication aggregation function shows far higher eclipsity as compared to the weighted linear sum aggregation function and thus may not be the most appropriate aggregation function for calculating the leachate pollution index. The variation in LPIwa values is comparatively sensitive and linear to changes in the subindex value of chromium and total iron throughout their range. Thus it can be concluded that the weighted linear sum aggregation function is the most appropriate aggregation function for calculating the leachate pollution index.

Table 5 Sensitivity Analysis Results of Selected Aggregation Functions for Change in Subindex Values of Chromium and Total Iron from 5 to 100

Number	Aggregation function	Changes in LPI values for chromium (%)	Changes in LPI values for total iron (%)
1	Weighted sum additive aggregation function, LPIwa	18.89	13.31
2	Weighted multiplicative aggregation function, LPIwm	21.18	14.48
3	Wighted root sum square aggregation function, LPIwr	13.76	0.09
4	Weighted 2.5th root (ambiguity and eclipsity free) aggregation function, LPIw2.5	11.99	8.63
5	Weighted 4th root sum aggregation function, LPIw4	8.73	6.35
6	Weighted 10th root sum aggregation function, LPIw10	4.42	3.25

CONCLUSIONS

Conclusion of this work are as follows**:**

[1] The unweighted linear sum, unweighted multiplicative, root sum square, root mean square, ambiguity and eclipsity free, fourth root, and tenth root sum aggregation functions are not suitable for aggregating subindices, as these aggregation functions do not take into consideration the importance / significance of all the variables, and assume that all the pollutant variables have some importance.

[2] The square root, ambiguity and eclipsity free (2.5th root), 4th root. And 10th root additive form aggregation functions also produce ambiguous results.

[3] The maximum operator aggregation function is ambiguity and eclipsity free, but it cannot be used as an aggregation function for LPI, as it is least sensitive to fine gradations of changes in leachate pollution.

[4] The two multiplicative aggregation functions (the unweighted aggregation function and the weighted aggregation function) produce highly eclipsed results.

[5] The weighted square root, weighted ambiguity and eclipsity free, weighted fourth root, and weighted tenth root aggregation functions are insensitive amd nonlinear to variations of individual pollutants.

[6] Though the weighted multiplicative aggregation function is most sensitive to the changes in pollutant concentration, is nonlinear and shows biased results for higher subindex values.

[7] Although the wighted linear sum does cover an underestimation region, it is less than that of the weighted multiplicative aggregation function.

[8] The weighted sum aggregation function is more parsimonious than the weighted multiplicative aggregation function.

[9] Hence, it can be concluded that the linear weighted sum aggregation method is the most suitable aggregation function for estimation of the most suitable aggregation function for estimation of the Leachate Pollution Index.

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Performance Study of Different Waste Made Organic Fertilizer Incombination With Chemical Fertilizers on Summer Bottle Gourd

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Abstract

The research was conducted with recommended dose of chemical fertilizer andcombination of different type of organic fertilizer with summer bottle gourd variety (BAR Lau-4)at the research field of HRC, BARI, Gazipur during Summer season of 2012. The experiment was laid out in Randomized Complete Block Design with three replication. The study was comprised of 11 treatments. The treatment combination weri.e T_1 = only chemical fertilizer (Urea, TSP, MP, gypsum and zinc sulphate @ 20000, 175, 175, 150, 100 and 12 kg/ha), T_2 = 3 ton city compost + 2/3 chemical fertilizer, T_3 = city compost + ½ chemical fertilizer, T4 =1.5 ton vermi compost + 2/3 chemical fertilizer, T_5 = 1.5 ton vermi compost + ½ chemical fertilizer, T_6 = 3 ton kitchen compost + 2/3 chemical fertilizer, T_7 = 3 ton kitchen compost + ½ chemical fertilizer, T_8= 3 ton city compost, T_9 = 1.5 ton vermi compost, T_{10} = 3 ton kitchen compost , T_{11} = control (no organic and inorganic fertilizer was used).The highest fruit yield (57.02 t/ha) was recorded where vermi compost alone with 2/3 of chemical fertilizer was used which was at per the yield of (56.17t/ha) where only chemical fertilizer wasused. Yield contributing other parameter were supported the yield. The highest BCR (1.8 Tk/Tk) was recorded where 1.5 ton/ha vermi compost alone with 2/3 chemical fertilizer was used compared with only chemical fertilizer and control treatment. Application of different type of organic compost alone with chemical fertilizer increases yield as well as BCR increases which is the indication of benefit. Actually safe vegetables production in is the major concern of the consumers.

***Key words**: organic waste, safe production, summer bottle gourd*

Introduction

Utilization of organic wastes grown in the city have the great potential to produce safe vegetables in the periurban areas of Bangladesh. Despite ongoing conversion towards more sustainable farming methods in developed countries and government support to further boost organic production, consumption of organic foods is expected to continue to outgrow domestic production in developed countries, leaving room for significant organic imports, at least in the short- to medium-term and probably beyond. Moreover, tropical and off-season products will continue to provide an attractive potential for which many developing countries have comparative advantages. The economies of many developing countries are dependent on the export of a relatively small number of agriculturalcommodities. As a result, diversification of agricultural production is more than ever of utmost importance. Diversification towards high-value crops can help to reduce the vulnerability of many agricultural producers in those countries, especially for resource poor and small scale farmers. During the second half of the 1990s, a strong and steady growth in the sales of organic foods has provided these products with a viable and sometimes value added market niche. Changes in dietary habits among many segments of the population of developed countries - resulting from increased health awareness and the increasing demand for a wider variety of products, including convenience food - have contributed to this growth.Bottle gourd (*Lagenariasiceraria*) is nutritious and a very popular vegetable in Bangladesh. It is widely cultivated throughout the country during winter season. It is cultivated during summer season in different parts of Bangladesh. A good number of high yielding varieties are available in Bangladesh. Our farmers are normally cultivate with chemical fertilizer. Yield is satisfactory level. But now-a-days, awareness are becoming about organic fertilizer. Farmers are interested to use different type of oragnic fertilizer. Moreover, they have become more demanding regarding information on production of the product. Sales of organic horticultural products have been expanding rapidly in many of the major organic markets e.g. the United States, countries in the European Community and Japan. However, the market share of organic products in total food sales becoming increasing.Keeping this in mind, different type of organic fertilizer alone or combination with chemical fertilizer were tested on vegetable. Yield almost at per like chemical fertilizer. Production cost is comparatively lower in case of using organic fertilizer. So, the study has been conducted for find out the suitable organic or organic with chemical fertilizer dose for bottle gourd production.

Materials and Method

The study was conducted at research field of HRC, BARI, Gazipur, during the summer season of 2012. Before starting the study soil sample was analyzed and noted down the different level of nutrient status. Summer bottle gourd variety BARI Lau-4 was used as planting material. The seeds of bottle gourd variety were sown in the polybag on 04 April 2012. Thirty days old seedlings were transplanted in the main field on 03 May, 2012. The experiment was laid out in Randomized Complete Block Design with three replication. The study was comprised of 11 treatments. The treatment combination weri.e T_1 = only chemical fertilizer (Urea, TSP, MP, gypsum and zinc sulphate @ 20000, 175, 175, 150, 100 and 12 kg/ha), T_2= 3 ton city compost + 2/3 chemical fertilizer, T_3 = city compost + ½ chemical fertilizer, T4 $_=$1.5 ton vermicompost + 2/3 chemical fertilizer, T_5 = 1.5 ton vermicompost + ½ chemical fertilizer, T_6 = 3 ton kitchen compost + 2/3 chemical fertilizer, T $_{7\ =}$ 3 ton kitchen compost + ½ chemical fertilizer, $T_{8=}$ 3 ton city compost, T_9 = 1.5 ton vermicompost, T_{10} = 3 ton kitchen compost , $T_{11\ =}$ control (no organic and inorganic fertilizer was used). The total amount of all chemical fertilizer except urea and MP, all organic compost were applied during final land preparation in the pit. Total urea and MP were applied in four equal installments at 21, 35, 55 and 75 days after transplanting. The intercultural operations were done as and when needed. Data on fruit length (cm), fruit diameter (cm), no. of fruits/ plant, single fruit wt. (kg), yield/plant (kg), yield (t/ha), fruit shape and fruit colour were recorded from three randomly selected plants per entry. The data on different characters was statistically analyzed.

Results and Discussion

The performances of bottle gourd during summer-rainy season are presented in the table1. The highest number of fruits (14.20) was recorded from the treatment (T_4)where 1.5t vermi compost used with 2/3 chemical fertilizers which number (13.60) was closed to recommended dose of chemical fertilizer. The lowest number of fruit (3.10) was recorded from control treatment. The biggest fruit (2.11 kg) was recorded from T_4 treatment. The highest length and breadth (38.93 cm and 13.01cm) was recorded from T_4 treatment. The highest per plant fruit weight (38.45 kg) was recorded from T_4 which closed to recommended chemical fertilizer dose. The highest fruit yield (57.02 t/ha) was recorded where 1.5t vermi compost and 2/3 chemical fertilizer were used which was at per the yield (56.17t/ha) of where recommended dose of chemical fertilizer was used. Yield contributing all parameter were supported the yield. The highest BCR (1.8 Tk/Tk) was recorded from the combination of 1.5t vermi compost and 2/3 recommended chemical fertilizer were used which was highest among all the treatment combination. The lowest BCR (-1.0 Tk/Tk) was obtained from control treatment.

Table: Yield and yield contributing parameter of summer bottle gourd variety (BARI Lau-4)

Treatment	No. of fruit	Individual fruit wt.(kg)	Fruit length (cm)	Fruit breadth (cm)	Fruit wt./plant (kg)	Fruit yield (t/ha)	BCR(Tk/Tk)
T_1	13.60	2.17	37.16	12.54	37.20	56.17	1.4
T_2	9.50	1.45	28.25	8.75	27.90	38.26	1.5
T_3	8.15	1.32	25.50	7.90	25.75	36.72	1.4
T_4	14.20	2.11	38.93	13.01	38.45	57.02	1.8
T_5	12.60	2.02	36.54	12.04	35.80	47.10	1.4
T_6	11.80	1.90	33.85	11.75	33.20	45.25	1.4
T_7	10.20	1.82	31.45	10.65	30.80	43.50	1.3
T_8	5.20	0.95	18.24	5.10	17.20	25.10	1.2
T_9	6.40	1.02	20.32	5.95	18.80	30.02	1.2
T_{10}	8.20	1.18	18.54	7.05	17.20	28.42	1.1
T_{11}	3.10	0.52	14.20	4.25	9.58	10.15	-1.0

Conclusion

It may be concluded that organic fertilizer alone with chemical fertilizer gave excellent response. Total production as well as BCR was highest compared to only chemical or combination with other organic fertilizer. Now a day, safe vegetables production in the country is the major concern of the consumers. Application of different compost along with chemical fertilizer increases the BCR through which growers may deposit the benefit.

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Effect of Organic Waste Separation Method and Use of Compost in Safe Production of High Value Summer Tomato and Bitter Gourd

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***Key words**: organic waste, source separation, safe vegetables, high value*

Vegetables are rich in vitamins and minerals and termed as protective food. Bangladesh is blessed of more than 100 vegetables. But the productions of vegetables are far below of the requirement. There exists widespread malnutrition in Bangladesh. Moreover, the modern cultivation techniques of vegetables are based on petroleum dependent synthetic fertilizers and pesticides. For getting higher yield the use of fertilizers and pesticides are being increasing day by day. As a result contaminated products are grown by farmers which cause different ailment instead of supplement the nutrients to human body. Therefore, consumers are preferring GAP product and organically grown safe vegetables in the country and abroad. Organic matter (OM) is the store house of nutrient in the soil and due to increased cropping intensity dependency on synthetic fertilizers, the OM of the Bangladesh soil is depleting and possesses far below amount compared to standard limit for having sustainable yield of crops (Karim 1999). Cattledung and plant debris's are the source of OM to add in the soil, but these sources are not adequate enough to meet the demand of the country.

There is another important source of organic matter, it is urban agricultural organic waste come from house hold, kitchen market, hotel and restaurant etc if we could utilize properly. It was documented that around 70% of the house hold waste is organic and everyday 500-1000g organic waste is produce in each house hold of Dhaka city. Approximately 13,332 tons of waste is produced per day in the urban areas of Bangladesh, which is over 4.86 million tons annually. It is projected that this amount will grow up to 47,000 tons/day and close to 17.2 million tons per year by 2025, due to growth both in population and the increase in per capita waste generation. Based on the present total urban population, per capita waste generation rate is found at 0.41 kg/capita/day in urban area (JICA 2005). But there have no effective steps to utilize this valuable potential resource which may use our crop field. Moreover due to lack of modern methods of decomposition, municipalities dump the waste in a landfill traditionally which is contributing climate change by emitting CO_2 and composted substance may have heavy metals. There are many ways to composts the waste aerobically and minimize CO_2 emission 70 % than landfill decomposition and produce good quality fertilizer may use in the safe vegetable production. Safe vegetables is termed vegetables having good quality, good condition and fresh, low in pesticide residues (below MRL Level), nitrate and heavy metal and free from pestilent insect and harmful microorganism. At present Bangladesh agriculture particularly vegetable production are mostly chemical fertilizer and pesticide based, which make the crop blemish with pesticide residue, nitrite contained due to use of excessive nitrogen. Therefore, safe vegetables become questionable to the end users. Therefore present study was done to standardize the dose of different compost along with chemical fertilizers which may use to grow safe vegetables under GAP.

Olericulture Division, HRC, BARI, Gazipur has initiated organic waste management program and composting facilities has developed in its research field. Initially traditional landfill composts were collected from municipal dumped site and tested it in the SRDI laboratory and later source level separation were made by providing three buckets to the BARI campus house holds. Three kinds of sticker also put on the bucket so that housewife may under stand which type of product has to put which bucket. Then the housewives were trained on separation method. Usually green vegetables waste and fruits peels were collected in one bucket for vermicomposting and left over cooked food and fish debris were suggested to kept in second bucket. All other wastes were requested to put in the third bucket. Every collection were made by the collectors. Vermicompost and aerobically produced compost were prepared in the vegetables field. Eleven treatment were constructed with the fully matured compost for each evaluation, these were, T1 = Recommended dose of Chemical fertilizers(RD), T2=City Compost 3t/ha+ 2/3rd RD, T3= City Compost 3t/ha + ½ RD Chemical fertilizer, T4= Vermi 1.5t/ha + 2/3rd RD , T5=Vermi 1.5t/ha + ½ RD, T6= Kitchen waste 3 t/ha + 2/3rd RD, T7=Kitchen waste 3t/ha + ½ RD, T8=City compost 3t/ha, T9= Vermi 1.5 t/ha, and T10=Kitchen waste 3t/ha. T11=No fertilizer. Randomised complete block design was followed with three replications for both experiments. The seeds of with BARI Hybrid Tomato 8 were sown in the seed bed on May 5, 2012. Seedlings were transplanted in the main field on May 30, 2012. The unit plot size was 3.6m x 2.3m accommodating 24 plants in each plot having 60cm x 40 cm plant spacing. The crop was fertilized with urea 550kg, TSP 450kg and MP 250kg per ha, respectively. Half of the quantity of compost, entire TSP and half of the MP were applied during land preparation. The remaining half of the compost was applied during pit preparation. The rest of MP and entire urea were applied in three equal installments at 15, 30 and 45 days after transplanting incompliance with treatment. The crop was protected from rain providing polythene tunnel. Irrigation, pruning, mulching weeding and other intercultural operations were done as and when necessary. For bitter gourd, seeds of BARI karola 1 were sown in the poly bag on 20 May 2012 and seedlings were transplanted in the main field on 8 June 2012. The experiment was laid out in RCB design with two replications. The unit plot size was 7.5 m x 1.2 m maintaining 50 cm drain and 1.5 m (plant to plant) spacing. The land was fertilized with cow dung, urea, TSP, MP, gypsum and zinc oxide @ 200, 175, 175, 150, 100 and 12 kg/ha, respectively. The total amount of compost, TSP, gypsum, zinc oxide and 1/3rd of each of urea and MP were applied in pit. The rest of MP was applied in two equal installments at 10 and 20 days after transplanting. Urea was applied in four equal installments at 21, 35, 55 and 75 days after transplanting incompliance with treatment. The intercultural operations and plant protection measures were done as and when needed. Data on days to 1st female flowering, node order of 1st female flower, marketable fruits per plant, single fruit wt.(g), fruit length (cm), fruit diameter (cm), yield per plant (kg), fruit shape and fruit color were recorded from three randomly selected plants per replication. Data were recorded for some yield and yield contributing characters. The compiled data were subjected to statistical analysis for interpretation of results.

Prevalence of heavy metal in the mixed are presented in the table 1 and revealed that mixed dumped compost contain huge amount of heavy metal. Similar results also observed Epstein *et al, 1992.* Application of these compost in the soil may contaminate soil and heavy metal come the food chain which may cause serious health hazard to the consumers.

Heavy metal	Mixed landfill compost (mg/kg)	Separated compost (mg/kg)
Cd	2.6	0.0
Cr	31.2	8
Pb	231	37
Ni	27	6
Zn	543	145

Yield and some yield contributing parameters of summer tomato and bitter gourd were presented in Table-2 & 3. Results clearly indicated that application of kitchen compost, vermicompost and city compost significantly influenced summer tomato production. Significance differences were found in earliness while applied the different compost along with chemical fertilizers. Individual fruits weight also differed statistically with recommended dose (RD) of fertilizer while compared with different compost treatments. Reduced size fruits have harvested while only applied compost, but good size was observed even 25% reduction of RD. TSS percent was influenced by the different compost. As higher TSS percent is desired during summer therefore, it is recommended for application vermicompost or kitchen for improve the quality of tomato. Yield per plant was observed similar at 25 % reduction of chemical fertilizers and BCR was observed the highest in those treatment.

Table-1. Yield and yield attributes of summer tomato variety BARI Hybrid Tomato 8

Treatment	Days to 1st harvest	No. of fruits / plant	Individual fruit wt. (g)	TSS (%)	Fruit yield/ plant (Kg)	BCR
T1	50	27	80.2	4.6	2.5	1.2
T2	46	26	80.2	4.9	2.6	1.8
T3	47	28	75.3	5.0	2.3	1.4
T4	46	27	79.3	5.0	2.4	1.8
T5	46	27	72.5	4.9	2.1	1.3
T6	47	30	80.1	5.0	2.5	1.8
T7	49	26	71.2	5.0	2.0	1.4
T8	49	27	73.3	4.9	2.0	1.2
T9	47	26	74.5	5.0	2.1	1.3
T10	50	28	70.6	5.0	1.9	1.2
T11	50	26	63.3	4.8	0.5	-1.3
F-test	*	Ns	**	*	*	

T1 = Recommended dose of Chemical fertilizers(RD), T2=City Compost 3t/ha+ 2/3rd RD, T3= City Compost 3t/ha + ½ RD Chemical fertilizer, T4= Vermi 1.5t/ha + 2/3rd RD , T5=Vermi 1.5t/ha + ½ RD, T6= Kitchen waste 3 t/ha + 2/3rd RD, T7=Kitchen waste 3t/ha + ½ RD, T8=City compost 3t/ha, T9= Vermi 1.5 t/ha, and T10=Kitchen waste 3t/ha. T11=No fertilizer

Table 3 demonstrated the yield and yield contributing characters of bitter gourd and revealed that application of kitchen compost, vermicompost and city compost significantly influenced bitter gourd production. Significance differences were found in earliness while applied the different compost along with chemical fertilizers. Individual fruits weight also differed statistically with recommended dose (RD) of fertilizer while compared with different compost treatments. Reduced size fruits have harvested while only applied compost, but good size was observed even 25% reduction of RD. Yield per plant was observed similar at 25 % reduction of chemical fertilizers and BCR was observed the highest in those treatment. It was due compost also contain some additional micro elements essentials for plant growth and development.

Table 2. Yield and yield contributing characters of bitter gourd variety BARI Korola 1

Treatment	Days to 1st harvest	No. of fruits / plant	Individual fruit wt. (g)	Fruit yield/ plant (Kg)	BCR
T1	45	13	80.2	4.7	1.3
T2	43	12	80.2	4.5	1.7
T3	45	14	75.3	4.2	1.4
T4	42	13	79.3	4.5	1.6
T5	46	12	72.5	4.1	1.3
T6	47	12	80.1	4.5	1.7
T7	49	13	71.2	3.8	1.3
T8	49	14	73.3	3.2	1.1
T9	47	12	74.5	3.5	1.2
T10	50	10	70.6	3.1	1.1
T11	50	7	63.3	1.1	-1.0
F-test	*	*	**	*	

T1 = Recommended dose of Chemical fertilizers(RD), T2=City Compost 3t/ha+ 2/3rd RD, T3= City Compost 3t/ha + ½ RD Chemical fertilizer, T4= Vermi 1.5t/ha + 2/3rd RD , T5=Vermi 1.5t/ha + ½ RD, T6= Kitchen waste 3 t/ha + 2/3rd RD, T7=Kitchen waste 3t/ha + ½ RD, T8=City compost 3t/ha, T9= Vermi 1.5 t/ha, and T10=Kitchen waste 3t/ha. T11=No fertilizer

Organic waste is a valuable materials produced everyday in our life. Proper separation and recycling method can make it more valuable to produce safe vegetables in the country which is the major concern of the consumers. Source level separation have some limitation but have good impact to get free the from heavy metal in the food chain. Moreover, application of different compost along with chemical fertilizer increases the BCR through which growers may deposit the benefit.

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Identification of Appropriate Landfill Sites for the City of Chittagong

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ABSTRACT

Urban solid waste management is a serious environmental issue confronting the cities in developing countries like Bangladesh. Apathy towards the problem, inadequacy of field level information and data, and resource constraints may be blamed for the dismal situation of solid waste management often visible in our cities. The most common problems associated with the absence of sustainable solid waste management practice include diseases transmission, odour nuisance, atmospheric and water pollution, visual blight, fire hazards and economic losses. In the three major cities of Bangladesh, Dhaka, Chittagong & Khulna, per capita production of solid waste is around 0.4kg /capita / day, but only a fraction of this waste is carried to the final disposal site. A recent study on Municipal Solid Waste Management, funded by Bangladesh Municipal Development Fund (BMDF), found that waste generated in Chittagong was 0.352kg/cap/day. Considering per capita generation of solid waste as 0.352kg to 0.4kg per capita per day, for a population of 25, 92,459 distributed within the 41 wards of the city (BBS 2011), total solid waste generated in Chittagong will be around 913 tons to 1037 tons per day in 2012.

Currently, Chittagong City Corporation has only two dumping yards: one at Ananda Bazar, Halishahar at the mid western part of the city and the other at Arefin Nagar, Pahartali at the northern tip of the city. None of these sites are sanitary landfill. Considering the city area of 168 sq.kms, only two dumping sites are not sufficient to cater to the requirement of the city. Long distances between the collection points and the disposal site are responsible for inefficient utilization of the CCC trucks and the resulting increase in the haulage time that eventually increases the costs of collection and disposal.

This paper aims to identify a number of appropriate landfill sites within the city for cost effective, efficient and safe disposal of solid wastes. An appropriate landfill site must satisfy environmental safety criteria and attributes that will enable the wastes to be isolated so that there is no risk to people and the environment. The criteria utilized for identifying appropriate landfill locations include natural physical characteristics as well as socioeconomic, ecological, engineering and land-use parameters. This study utilized Geographic Information System (GIS) and Multi-Criteria Decision Method (MCDM) for the identification and selection of appropriate landfill sites within the city of Chittagong. Thirteen sites were identified initially. Out of these sites four were found to be most appropriate and suitable for use as landfill. The sites thus identified, if utilized by CCC, are expected to bring a major change, in terms of cost effectiveness, efficiency and safety, in the current scenario of solid waste management in the city.

Keywords: Solid Waste Management, Sustainable, Geographic Information System, Multi-Criteria Decision Method, Chittagong, Bangladesh

INTRODUCTION

Background information

Bangladesh is one of the most densely populated countries in the world having a population of 965 per square km (BBS 2011). Rapid urbanization has also become a visible feature, especially in the three major cities of Bangladesh: Dhaka, Chittagong & Khulna, after the liberation of Bangladesh in 1971. While the total population of the country has been increasing at 1.37% per annum (BBS 2011), the urban population is growing at about 3.27% per annum (Aqua Consultant et.al 2012). This rapid urbanization has created a strain on the resources of local bodies like cities, towns and municipalities, which are often finding it difficult to cope with the increasing demands of the city dwellers for urban services and civic amenities. Solid waste management is one such area of urban services where funds are often in short supply; always giving way to other pressing needs like health care and education. Chittagong is the second largest city in Bangladesh having a population of about five million including people living in the urban fringes. Managing solid waste efficiently and arranging resources for it has become a serious concern of the city authority. Currently, Chittagong City Corporation has only two dumping yards: one at Ananda Bazar, Halishahar at the mid western part of the city and the other at Arefin Nagar, Pahartali at the northern tip of the city. None of these sites are sanitary landfill.

Problem statement:

Chittagong is the second largest city of Bangladesh. According to the preliminary results of the population and housing census 2011, the city has a population of 25, 92,459 distributed within the 41 wards of the city (BBS 2011). The city is growing. It grew at the rate of 4.527% per annum between the years 1991 -2001. However, it appears, the growth has slowed down between the years 2001 to 2011. The city grew at the rate of 2.81% per annum in average during the last decade (BBS 1997, 2002, 2011). Sujauddin reported a per capita generation of 0.25kg of solid waste in Chittagong based on their field survey and analysis of solid wastes generated in a small community in Chittagong city (Sujauddin, M et al 2007). Compared to other similar cities in South Asia, this per capita generation of solid waste in Chittagong appears to be very low. Inferring from the waste generation in similar cities in South Asia, Ashraf estimated a per capita per day generation of 0.4kg of solid wastes in Chittagong (Ashraf 1994). A recent study on Municipal Solid Waste Management, funded by Bangladesh Municipal Development Fund (BMDF), found that waste generated in Chittagong was 0.352kg/cap/day. This finding by BMDF appears to be more realistic. Therefore, considering per capita generation of solid waste as 0.352kg to 0.4kg per day, total solid waste generated in Chittagong will be around 913 tons to 1037 tons per day in 2012. Mr.Shafiqul Mannan, Chief Conservancy Officer, Chittagong City Corporation (CCC) claims that nearly 90% of the waste generated in Chittagong are collected and disposed of by them. Currently, they are using 60 trucks with a varying capacity of three to five tons for collection and disposal of solid wastes within the city. These trucks can give only three to four trips per day: two trips during daytime and two trips during the night. These trucks could give a maximum of 240 to 250 trips, according to him, in a day (Mannan 2012). For an efficient and sustainable solid waste management system, these trucks should be better utilized giving more trips per day than what it is giving now. Considering the hauling distances covered by the waste trucks and the travel time required, dependence on these two sites only, eccentrically located at the northern and mid western end of the city, does not seem to be justified on economical consideration. Several sanitary landfill sites at appropriate location evenly distributed within the city might be the answer to the problem outlined above.

AIMS AND OBJECTIVES

Identifying appropriate sites to be used as sanitary landfill within the city area is the aim of this study. If the current city area of 168 sq. km is sub-divided into four quadrants and at the least four appropriate sites, each for one of the quadrants, could be identified for using as sanitary landfill, efficiency of solid waste collection and disposal could be greatly increased. Haulage distance and time for city corporation garbage trucks will be greatly reduced. Garbage trucks will be able to give more trips as against the current practice of three to four trips per day by each truck. This paper aims to identify appropriate landfill sites within the city by using geographic information system and multi criteria decision analysis. In course of the decision making process, environmental, economical and ecological factors will be taken into consideration.

STUDY AREA

For analysing the problems and issues relating to solid waste management in the City, we have taken the City of Chittagong in its current setting as our study area. It is the second largest city of Bangladesh and has an area of about 168 sq.km. It is located between 22°-14´ and 22°-24´-30´´ N Latitude and 91°-46´ and 91°-53´ E Longitude on the Right Bank of the river Karnaphuli. As per census 2011, the city has a population of 25, 92,439 people. Considering the urban fringes, estimated population of the city will be around five million. The study area is illustrated in Fig.1.

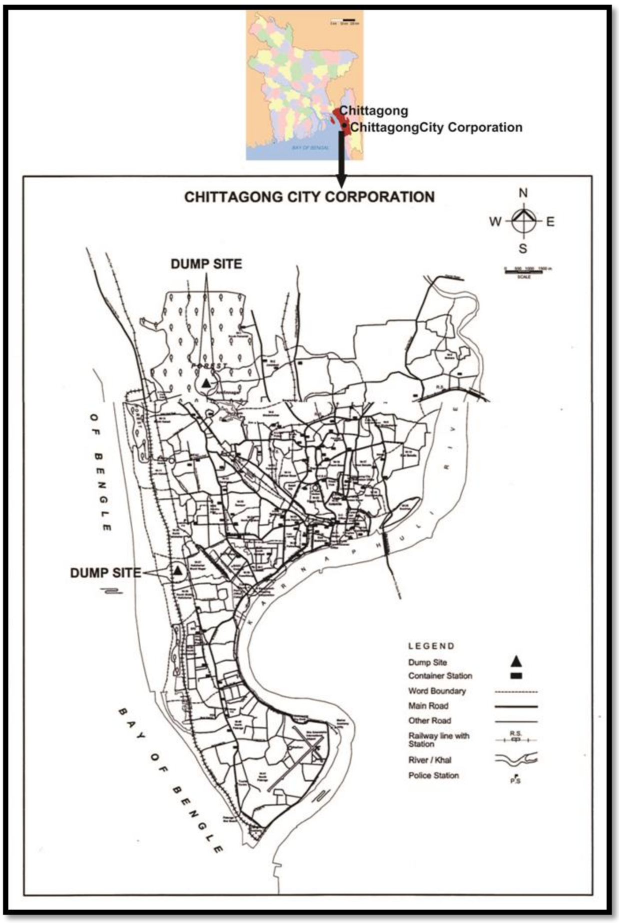

Fig 1: Chittagong City Corporation (CCC)

PRESENT SCENARIO OF SOLID WASTE COLLECTION IN THE CITY AREA

Current capacity of the Chittagong City Corporation:

At present there are two dumping yards at Arefin Nagar, Pahartali and Anandabazar, Halishahar. Arefin Nagar yard has an area of 73 acres while Halishahar yard is smaller in size and has an area of ten acres. The dumping yards have no provision of leachate collection. Assuming weight of solid waste as 500kg per cubic meter and allowable height of fill as four meter, we will need around 42 to 46 acres of land in Chittagong, to be used as landfill, per year at the present generation of around 913 tons to 1037 tons of solid waste per day. If we assume, conservatively, five years will be necessary to convert and remove at least 60% of the garbage as compost from the landfill site, the city will need around 322 acres of landfill at the present stage to begin with. Land requirement will be more if we have to accommodate the increased generation over the years because of population growth. Amongst the total wastes disposed of in Chittagong, 71.7% is organic and 28.3% is inorganic waste at household level. Currently, 12.41% of all wastes generated are recycled and only 1% is composted (Aqua consultant et.al 2012). According to the information given by the Chief Conservancy Officer, CCC, 50% of the solid wastes generated goes to Anandabazar for disposal and the remaining 50% goes to Arefin Nagar at Pahartali. CCC has a fleet of 84 trucks out of which 60 trucks are in working condition (Mannan 2012). Open fixed bed truck are expected to make minimum 3 trips per day while dump trucks are expected to make 4 trips per day and container trucks are expected to make between 6-8 trips per day. According to load carrying capacity of the trucks and number of trips they can make in a single day, total waste carrying capacity of the existing waste collection fleet of CCC is estimated at 830 tons per day. There are 507 rickshaw vans used for garbage collection from the households in different city wards. The number of trips made by each rickshaw van varies between 2 to 4 trips and wastes carried vary between 350 kg to 400 kg. There are 1269 dustbins and demountable containers at 76 locations within the city (Aqua Consultant et.al 2012).

Limitations of the existing dumping yards within the city

Table 5: Limitations of the existing dumping yards in the city

Site name	**Specific location**	**Buffer zone-Separator**	**Soil cover**	**Protection against rain**	**Leachate treatment**	**Gas monitoring system**	**Accessibility**	**Physical condition**	**Environmental acceptability**
Arefin Nagar	Asian women university is located within 100m	No	No	No	No	No	Not good	Poor, leachate is flowing and has chance to mix with surface water	Odour, pollution and noise are found, no incineration. No social acceptance
Halishahar	RCC protect. wall along the sea shore within 10 meter	No	No	No	No	No	Good	Mixing of medical waste, cattle grazing is common, vehicle moves across the land fill	Odour, pollution and noise are found. Waste is incinerated. Not socially accepted

Case study: lessons from Sugandha Residential Area

Sugandha Residential Area is one of the posh residential areas in Chittagong. During the Pakistan period, it was a dumping yard for solid wastes. After relocation of dumping sites to Yakub Nagar & Halishahar, CCC turned the site into a residential area. Several residential plots were created and sold. The plot owners while building structure on it faced problem because their structural engineers did not allow them to place foundation on this decompose-able fill. They either had to remove all the fill materials or go for costly cast in situ piles in foundation. Double expenses in the form of public and private expenditure were incurred in filling the area once and then removing the fill again. If the dumping site or landfill site is selected as a part of a conscious land use plan, the site could be used as a park, playground, botanical garden or alike. This could have saved the inconvenience and cost of excavating and removing the fill for laying foundation. If we can consciously identify landfill sites, keeping in mind that these sites will be eventually used as a playground, park, open space, vegetable garden, botanical garden, green corridor etc., city will be benefited. After using a landfill for fifteen or twenty years, the site can be abandoned as a landfill and can be converted into a playground or park at a reasonable cost.

METHODOLOGY

Primary data from the field trips, reconnaissance survey, literature survey, key person interview, newspaper reports, web sites etc. were the main source of crude data and information. GIS Maps produced by Chittagong Development Authority (CDA) including Physical Feature Survey Maps and the Contour Maps, Structure Plan, Transportation & Drainage Master Plan, Detail Area Plan for Chittagong and photography were extensively used in analysing, processing and recording the data and information collected. Two types of analysis were done: GIS based analysis and Multi Criteria Decision (MCD) based analysis.

*GIS based analysis**:*** To conduct this analysis, Geographic Information System (GIS) software was used. As a first step, different buffer zones were created to separate residential, commercial, community facilities, health facilities, educational institution, pucca access roads, agricultural land etc. from the proposed landfill sites. The following table indicates the minimum buffer distances kept between the landfill sites and the existing land usages while identifying the appropriate landfill sites.

Table-1. Minimum buffer distances from the proposed landfill sites considered for different land usages in this study

Description	**Minimum Distance from proposed landfill**
Residential	300 meter
Commercial	300 meter
Community facilities like bazaar	500 meter
Educational Institutions	500 meter
Health Facilities	500 meter
Water bodies like khals and rivers	300 meter
Ponds over the size of 0.5 acres	300 meter
Pucca access roads	200 meter
Agricultural land	200 meter

Digital Elevation Model (DEM) was prepared from contour map to determine the land characteristics.

MCD based analysis: This step involves analyzing the suitability of the landfill sites on the basis of criteria like hydrological characteristics, climate, wind direction, underground soil condition, socio economic parameters, distance from the water body, land price, accessibility etc. Detail Area Plan (DAP) for Chittagong has barred filling of any pond over the size of 0.5 acres. Therefore, all endeavours were made to locate the landfill sites, at the least, 200 meter away from the ponds over the size of 0.5 acres. To prevent leachate reaching the water table information regarding soil characteristics were analyzed. Landfills with clay layer underneath were considered highly suitable and tagged very high. A study with soil in Khulna found sub soils in four selected locations in and around Khulna City suitable as a compacted clay liner for the construction of sanitary landfill (S.Akhter et al 2008). Considering the suitability as sub soil, silty clay is tagged high while mixed silty or sandy clay is tagged moderate and sandy soil: low respectively.

RATIONALE BEHIND THE STUDY

Chittagong City Corporation covers an area of 168 sq km generating 913 to 1037 tons of waste per day. It is facing difficulty in coping with the problems of solid waste collection and disposal. Its present system of solid waste collection and disposal is not efficient and cost effective. With the gradual increase in population, the situation is going to deteriorate even further. The city with its existing manpower and expertise is not in a position to find a way out of this problem. The present study has tried to pin point the problem and its magnitude. It has tried to give a solution, too. The city might consider implementation of the solution proposed.

FINDINGS

After compiling all buffer layers and applying multi criteria decision analysis, this study has identified thirteen primary locations falling within the four quadrants of the city area. Characteristics of these sites are given below:

Table 2: Characteristics of primarily identified landfill sites

Site no.	Location	Specific Location	Elevation	Soil suitability based on bore log analysis	Land Price per Katha)	Land size	Accessibility	Remarks
1	South of the city in South Patenga, ward-41	Near Naval Academy	Low land 3.5m above mean sea level	Bore log not available	5 lacs	-------	Good	Satisfies most of the controlling criteria. There is a small pond and a khal passes through the area
2	South of the city in South Patenga, Ward-41	Beside Patenga CNG Station.	Low land 3m above mean sea level	Bore log not available	5 lacs	--------	Good	Satisfies most of the controlling criteria. Near char land (shoal) There are several small water bodies over the size of 0.5 acres
3	South of the city at South Halishahar, Ward-39	Near CEPZ – Approx. 1km from BRAC Primary School.	Low land 2.5m above mean sea level	Moderate	8 Lacs	About 80 acres	Good	Satisfies most of the controlling criteria. Near char land (Shoal)
4	South of the city at South Patenga, Ward-41)	Situated at Taltola, Bandartila	Low land 2.7m above mean sea level	Bore log not available	----------	--------	Good	Satisfies most of the controlling criteria. One khal divides the area. Near land under port use

5	South of the city at South Patenga, Ward-41)	Situated at Hosain Ahmad Para, near airport.	Low land 3.5m from mean sea level	Bore log not available	----------	--------	Good	Satisfies most of the controlling criteria. Rail line crosses the area. One ditch and a pond greater than 0.5 acre in size exist.
6	West of the city at North Middle Halishahar, Ward-37+North Halishahar, Ward-26+ South Kattali, Ward-11+ North Kattali, Ward-10)	About one km from Chittagong Polytechnic for Girls.	Low land 2.5m above mean sea level	Moderate	5 lacs	About 160 acres	Good	Satisfies most of the controlling criteria. Several types of water bodies were found scattered in this area.
7	East of the city at Boxirhat, ward-35+ East Bakalia, Ward-18)	About 2km away from the east of Little Bird K.G school.	Low land 2.5m above mean sea level	Very high	----------	--------	Moderate	Satisfies most of the controlling criteria. Safe distance away from the river. One khal has entry to the site.
8	East of the city at Mohra, Ward-5)	About 1.5 km away from the east of Hamidpur Primary School.	Low land 2.5m above mean sea level	Very high	8 Lacs	About 50 acres	Good	Satisfies most of the controlling criteria. Safe distance away from the river
9	North Eastern part of the city at Mohra, Ward-5	About 1.5 km away from North-East of Chowdhury Bill Govt.Primary School.	Low land 3.5m above mean sea level	Very high	-----------	--------	Good	Satisfies most of the controlling criteria. A number of ponds >.5 acres in size were seen around the site
10	North	About	Low	High	----------	--------	Good	Satisfies all

	Eastern part of the city at Chandgaon,ward-4	100m away from CDA Anannya Residential Area	land 3m above mean sea level		-			controlling criteria.
11	North of the city at North Pahartali, Ward-9)	Approx. 500m away from West of Darul Quran Madrasa	Moderately high land 7m above mean sea level	Moderate	15 lacs	About 110 acres	Good	Satisfies all controlling criteria. Situated in hilly region Slums are existing
12	North of the city at Jalalabad,Ward-2)	Just near the cantonment area	Moderately high land 6.5m from mean sea level	Very high	----------		Good	Satisfies all controlling criteria.
13	North of the city at South Pahartali, Ward-1)	About 2km from the west of Chittagong University	High land 16m above mean sea level	Bore log not available	-----------		Good	Satisfies most of the controlling criteria. Situated in the hilly region -One Khal exist.

Source: Field information, collection and analysis 2012

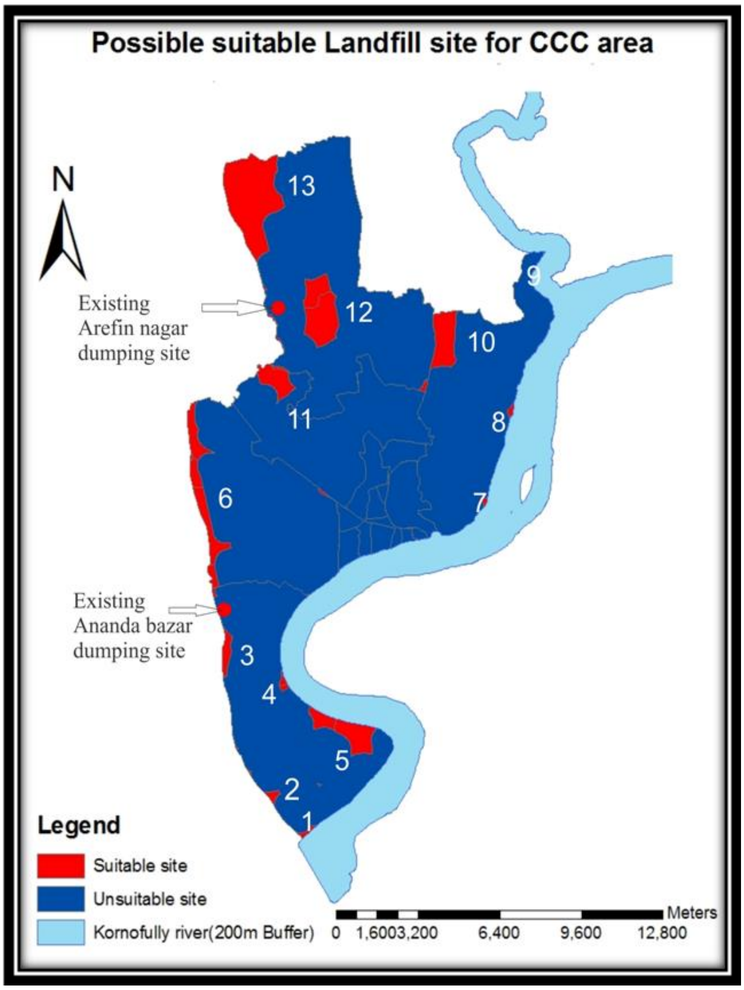

Fig 2: Possible landfill sites within the city

Appropriate sites

Among the 13 possible suitable sites identified for establishing sanitary landfills in the city, four were found most suitable. Site no 3 is located in South Halishahar, Ward No.39. It can cater to the requirement of the southern part of the city especially Patenga and part of Halishahar. Site no 6 spreads between four wards namely: North Middle HaliShahar-Ward 37, North Halishahar-Ward 26, South Kattaly-Ward 11 and North Kattali-Ward 10. It can serve the western part of the city area. Site No.11 at North Pahartali-Ward 9 can cater to the requirement of northern part of the city. Site no.8 is in Mohara-Ward 5. It will be most suitable for serving areas located in the east of the city. The aforementioned sites, considered suitable as a landfill, are illustrated in Figure-2.

Characteristics of the four most appropriate sites

Site no 3: The site is located approximately 1 km away from BRAC primary school in South Halishahar. It satisfies the entire buffer requirements elaborated in Table-1. It is located in a low-lying area and its elevation is 2.5m above mean sea level. Sub soil is mixed silty or sandy clay. Considering the possibility of infiltration of leachate into the groundwater table, the site can be termed moderately suitable as a landfill. The site is accessible from a pucca road. Accessibility can be termed as good. The total area of the site is about 80 acres. Land price is around Taka eight lacs per katha which is equivalent to Taka 49 million per acre. Soil characteristics of the site is given in Table-3

Table-3. Soil Characteristics of Site No. 3

Ward No	39	
Borehole Location	CEPZ	
Depth (m)	**Lithology**	SPT N Value
0-1.5	Silty sand	2
1.5-4.5	Silty sand with clay	8,11
4.5-10.5	Medium sand with clayey silt	11,17
10.5-19.5	Medium sand with clayey silt	18

Source: Soil quality analysis, 2012

The site is in zone-DPZ-3 of the Detailed Area Plan (DAP) for Chittagong published in the year 2008. It is located near the sea and the zoning provisions in DPZ-3 earmarked it as a tourist spot. After using the site for a number of years as a landfill, it can be converted into a tourist spot with landscaping, eateries, amusement park etc.

Site no 6: The site is located approximately one km away from west of Chittagong Girls Polytechnic. It satisfies the entire buffer requirements elaborated in Table-1. It is located in a low-lying area and its elevation is 2.5m above mean sea level. Sub soil is mixed silty or sandy clay. Considering the possibility of infiltration of leachate into the groundwater table, the site can be termed moderately suitable as a landfill. The site is accessible from a pucca road. Accessibility can be termed as good. It is a low-lying area: 2.5m above mean sea level. Land price of this area is about five lacs per Katha which is equivalent to Taka 30 million per acre. The site is about 160 acres in size. Soil characteristics of the site is given in Table-4

Table-4. Soil Characteristics of Site No. 6

Ward No	37	
Borehole Location	Wapda circle	
Depth (m)	**Lithology**	**SPT N Value**
0-1.5	Medium sand	7
1.5-3	Medium sand with clay silt	5
3-10.5	Inorganic soil with silty clay	13
10.5-13.5	Stiff silty clay	16

Source: Soil quality analysis, 2012

The site is in zone-DPZ-2 of the Detailed Area Plan (DAP) for Chittagong. The zoning provision allows Children park or alike in the area. After its use as a landfill, the site can be developed as a children park with space allocation for open-air concert, football ground etc. It can also be developed into a recreational area.

Site no 8: The site is located about 1.5 km away from Hamidpur Primary School in Ward 5, Mohara. It satisfies the entire buffer requirements elaborated in Table-1. It is located in a low-lying area and its elevation is 2.5m above mean sea level. At a depth of 1.5m from the ground level, the site has a grey soft to medium stiff inorganic clay of low to medium plasticity. This clay layer extends up to the depth of about 18m. The sub soil with a clay layer, about 16m thick, can be considered highly suitable for preventing leachate contaminating the ground water. Access is from a pucca road. Accessibility can be termed as good. Land price is around Taka eight lacs per katha equivalent to about Taka 49 million per acre. The site is about 50 acres in size. Soil characteristics of the area is given in Table-5

Table-5. Soil Characteristics of Site No. 8

Ward No	5	
Borehole Location	Nazir Ali Tendal Road, Mohora, Chandgaon	
Depth (m)	**Lithology**	SPT N Value
0-1.5	Light grey soft inorganic silty clay of low to medium plasticity	3
1.5-5.5	Grey soft to medium stiff inorganic clay of low to medium plasticity	4,5
5.5-18.5	Grey very soft inorganic clay silt of low plasticity	1,4

Source: Soil quality analysis, 2012

The site falls within zone DPZ-4 of DAP 2008. It is located near the river Karnafuli. DAP has proposed establishment of leisure centre, riverside promenade, picnic spot etc. in this area. After its use as a landfill, the site can be used for land usages proposed in DAP.

Site no 11: The site is located approximately 500 m away from the west of Darul Quran Madrasa. It satisfies the entire buffer requirements elaborated in Table-1. It is located in a hilly area having elevation 7 m above mean sea level. At a depth of about 3m from the ground level, the site has a grey soft to very soft clay layer extending up to a depth of about 12 m. The sub soil with a clay layer, about 10m thick, can be considered highly suitable for preventing leachate contaminating the ground water. Another soft clay layer starts from a depth of 17m and extends up to 20m. Accessibility is from pucca road and can be termed as good. Land price is around Taka 15 lacs per katha which is equivalent to about Taka 91 million per acre. The site has an area of about 110 acres. Soil characteristics of the site is given in Table-6

Table-6. Soil Characteristics of Site No. 11

Ward No	9	
Borehole Location	Near Panjabi Lane Foy's Lake Area	
Depth (m)	**Lithology**	**SPT N Value**
0-2.74	Brown filing sand	2
2.74-12.8	Grey, soft to very soft clay	2
12.8-16.77	Grey, Medium dense, silty sand	12,14
16.77-19.82	Grey soft clay	2

Source: Soil quality analysis, 2012

Site no 11 falls in Zone-DPZ-5 of the Detailed Area Plan (DAP). It is located in a hilly area. DAP wants it to be preserved. In the 1961 Master Plan, a botanical garden was proposed in that hilly region. After its use as a landfill, the site can be effectively developed into a botanical garden. A botanical garden in the northern part of the city might become its lung in future. In fact, even now, Chittagong does not have a botanical garden.

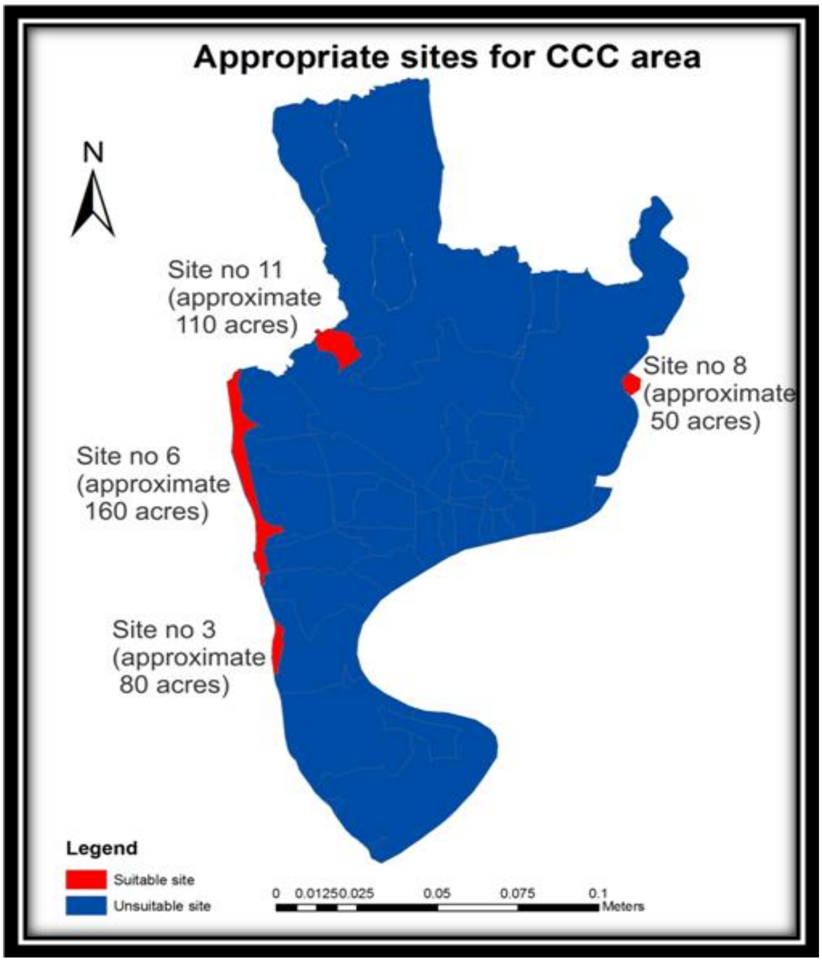

Fig 3: Most Appropriate Landfill Sites

ENVIRONMENTAL CONSIDERATIONS CONCERNING THE LANDFILLS

Site no 3, 6, and 8 are located in a low-lying area. Site no 11 is located in a hilly area. All these areas might encounter problem with leachate during the rainy season. The following procedures could be adopted to mitigate the problem. Schematic representation of a sanitary landfill with design components is shown in Fig.4.

- Synthetic liners including plastic geo-membranes, geo-mats, geo-grids, geo-textiles that commonly contain bentonite clays could be used to trap leachate. Synthetic liners in double layers could be used underneath the fill for creating a leachate trap. Leachate thus accumulated can be removed subsequently by using leachate collection pipes (Sener 2004).

- Cell system could be introduced to prevent infiltration of rainwater. The cell thickness may range from 8 to 30 feet (2.4 to 9.1m) but 15 feet (4.6m) thickness is commonly used. The width of the working face is limited to 2 feet (.6m). The first lift of the waste should be 5 feet (1.5m) or less with careful removal of the oversize pieces to prevent damage to the underlying leachate collection system. The thickness of the daily cover should be 6 to 12 inches (159 to 300mm). If the lift surface is anticipated to remain exposed for more than 30 days, an intermediate cover having a thickness of one foot should be used. This one feet thickness of cover will be more resistant to erosion compared to the usual daily cover (Sener 2004). When the operational life of the landfill is over, a final layer of soil and optimum synthetic liners should be used on top along with a vegetative cover to limit percolation and erosion.

- As several types of water body exists in site no 6, water body with less than 0 .5 acre in size may be filled considering the scarcity of land. Water bodies above the size of 0.5 acres should be preserved. A schematic representation of a sanitary landfill is shown in Figure-4.

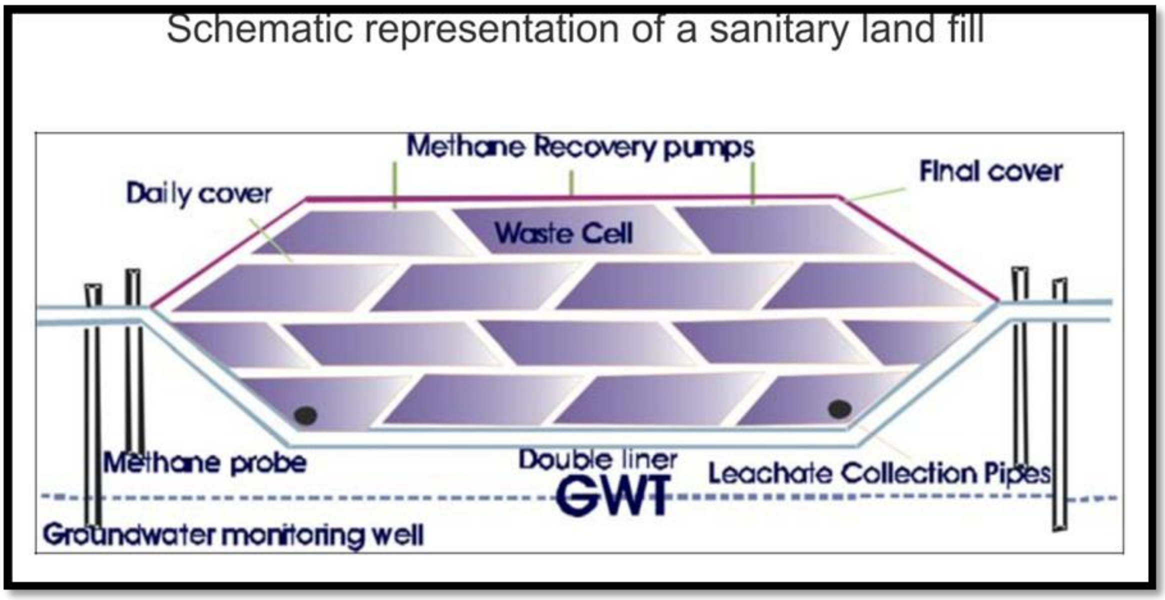

Fig 4. Schematic representations of a sanitary land fill with design components

Source: Tchobanoglous *et al.*,1993; Bagchi,1994

- Chittagong is a populous city. High-density development may take place around the landfills in future. A twenty-meter buffer zone with trees and shrubs is to be created surrounding the landfill. This buffer zone will help reduce odour and at the same time create a green visual barrier between other usages of land. Since wind blows generally form the southwest direction, there must be a green barrier on the eastern side of Site- 6 and Site- 3.

LIMITATION OF THE STUDY

- Soil characteristics mentioned in this study will be indicative in nature. Information based on one boring result will not be sufficient to justify a site as more suitable or less suitable as a landfill considering the permeability as a controlling parameter. More bore log results, distributed throughout a landfill, are to be analyzed; hydraulic conductivity of the sub soil-clay layer is to be assessed. Hydraulic conductivity should be around: k= 10^{-7} cm per sec to be effective as a soil liner. Chemical composition of the subsoil was not analyzed. Chemical composition of subsoil is an important parameter controlling the permeability of the subsoil underneath the fills. For projects needing huge investment, these analyses should be done before arriving at a final conclusion on appropriate landfill site.

- Not much information was available on area specific ground water condition.

- An average value of land price was given. Price varies depending on the location. Price is always high near the access road. Information regarding price was collected by interviewing local people only.

CONCLUSION

The findings of this study may give essential inputs to the policy makers in Chittagong City Corporation. They may use the information provided for identifying appropriate landfill locations for the city. The current dependence of the city on only two dumping yards, eccentrically located on the western side of the city, is not cost effective and therefore a revision in their waste disposal strategy is necessary.

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Effect of Different Compost in Combination with Chemical Fertilizer on the Safe Production of Okra

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ABSTRACT

The experiment was conducted with different compost in combination with chemical fertilizer for safe production of okra at Olericulture Division, Horticulture Research Centre, Bangladesh Agricultural Research Institute, Gazipur during the summer season of 2012. There was significant different on yield and yield contributing characters to different fertilizer combinations. Highest fruit yield per hectare (13.74) was produced by T_4 (Vermicompost 1.5t/ha + 2/3rd recommended doses of chemical fertilizer) followed by T_5 (Organic waste compost 3t/ha+ 2/3rd recommended doses of chemical fertilizer), T_2(Vermicompost 1.5t/ha + ½ recommended doses of chemical fertilizer) and T_3(Organic waste compost 3t/ha + ½ recommended doses of chemical fertilizer). The lowest fruit yield was produced by T_8 (Control) (4.85 t/ha). Vermicompost 1.5t/ha along with 2/3rd recommended doses of chemical fertilizer may be used for higher yield and safe production of okra.

Key words: Organic waste compost, vermicompost, safe production, okra

Introduction

Okra (*Abelmoschus esculentus* L. Moench.) is an important vegetable crop of Bangladesh belonging to the family Malvaceae, It is originated in tropical Africa (Purseglove, 1987) grown as a popular vegetable throughout the tropical and sub-tropical regions of the world. About 24,230 metric tons of okra is produced from 10204.00 hectares of land per year in Bangladesh, and its average yield is about 4.15 t/ha (BBS, 2011). Through it is grown round the year; its production is mainly concentrated during summer season. During rainy season, scarcity of vegetable is a problem in Bangladesh. Okra contributes a lot as a nutritious vegetable during lean period of vegetables. The soils of Bangladesh are fertile but the organic matters of soil are decreasing day by day. The organic matters of Bangladeshi soils are ranging now 0.5 to 1.00, but it is very alarming. Organic matter of soil is the heart of soil. Generally, the farmers are used cowdung as the source of organic matter but scarcity of cowdung is the very common in Bangladesh. Usually organic waste is piled in the open land and create hazard to the city dwellers. It also produces leachate during natural composting which pollute the soil environment. But organic waste is a very good source of organic fertilizer and produced good quality organic fertilizer. There is huge scope for utilizing city or kitchen waste convert to organic fertilizer. Organic waste compost and vermicompost were produced from kitchen organic waste which was used in the experiment. If we use organic waste compost in the agricultural field, it will be added organic matter in the soil as well as pressure of chemical fertilizer may be decrease and safe vegetable may be produced. Considering the above facts, the present research was undertaken to see the effect of different organic waste compost along with chemical fertilizer on safe okra production.

Material and Methods

The experiment was carried out during the summer season of 2012 at Olericulture Division, Horticulture Research Centre, Gazipur and Agricultural Research Station, Rangpur, Bangladesh Agricultural Research Institute from May to August, 2012. BARI Dherosh 1 was used for fruit yield as high yielding variety. The experiment was laid out in RCB design with 3 replications in the 4.0m x1.0m plot. Forty seeds were sown for each lines May 25, 2012 with the spacing of 60 × 40 cm. Manures and fertilizer were applied as recommended doses as: Urea 160kg/ha(73.0 kgN), Triple Super Phosphate(TSP) 150kg/ha(30.0 kg P) and Murate of Potash(MP) 150kg/ha(75.0 kg K)(Anon., 2005). The treatment combination were arranged as follows: T_1= Recommended amount of chemical fertilizers (RD), T_2= Vermicompost 1.5t/ha + ½ RD, T_3= Organic waste compost 3t/ha + ½ RD, T_4= Vermicompost 1.5t/ha + 2/3rd RD, T_5= Organic waste compost 3t/ha+ 2/3rd RD, T_6= Vermicompost 1.5 t/ha, T_7= Organic waste compost 3t/ha and T_8=Control (no fertilizer). The entire amount of organic fertilizer, TSP and 2/3 of MP were applied at the time of final land preparation. The remaining MP was applied after 30 days of seed sowing. Urea was applied in 3 equal installments at 25, 40 and 55 days after sowing. Weeding and other intercultural practices were done as and when necessary. No chemical measures were taken for pest infestation. Data were taken on days to 50% flowering, fruit size, fruit weight, number of fruits per plant, yield per plant, number of seeds per fruit, number of ridges per fruit, fruit yield per hectare. Data were analyzed statistically (Gomez and Gomez, 1983).

Results and Discussion

Results of yield and yield contributing characters of okra were presented in table 1. All the characters under the study except diameter of fruit and ridges per fruit differed significantly. The results revealed that days to 50% flowering were highly significant among the treatments. It ranged from 41.46 to 48.00. Number of branches per plant varied significantly and ranged from 2.00 to 3.76. T_4 produced higher number of branches per plant followed by T_5(3.70), T_1(3.70) and T_6(3.36). Internodes per plant differed significantly among the treatments and it ranged from 14.80 to 24.13. Plant height at last harvest was highly significant and tallest plant was found from T_4(1.20m) and followed by T_3(1.16m) and T_2(1.13m). For individual fruit weight, there had a significant difference; heaviest fruit was found from T_4(15.63g) followed by T_3(15.00g) and T_5(14.30g). Number of fruits per plant is very important character for higher yield of a crop; it varied significantly and ranged from 12.00 to 25.86 in different treatment combinations. Highest number of fruit was produced by T_4(25.86) and followed by T_5(22.00), T_2(22.00) and T_3(20.00). Fruit length differed significantly, longest fruit was found from T_5(16.25cm) and lowest in T_8(12.90cm). Diameter of fruit did not varied significantly and it ranged from 1.34 to 1.52cm. Number of seeds per fruit differed significantly and maximum seeds per fruit were found from T_5(29.50). There was no significant different in number of ridges per fruit and it ranged from 5.35 to 5.80. Fruit yield per plant showed highly significant, highest fruit yield per plant was produced by T_4(404.19g) and followed by T_5(314.60g), T_2(305.80g) and T_3(300.00g). The lowest fruit yield per plant was produced by T_8(142.80g). Fruit yield ton per hectare varied significantly and it ranged from 4.85 to 13.74. Highest fruit yield was produced by T_4(13.74t/ha) and then T_5(10.69t/ha), T_2 (10.39t/ha)and T_3(10.20t/ha). Minimum fruit yield per hectare was produced by T_8(4.85t/ha).

Considering the yield and yield contributing characters, it is concluded that application of vermicompost 1.5t/ha with 2/3rd of recommended dose of chemical fertilizer produced better yield of okra in relation to other fertilizer combinations. Therefore, it is recommended that vermicompost produced from kitchen organic waste is a potential organic fertilizer may be used for higher yield and safe cultivation of okra.

Table 1. Yield and yield contributing characteristics of okra

Treatments	Days to 50% flowering	No .of branches/ plant	No.of internodes/ plant	Plant height at last harvest (m)	Individual fruit weight (g)	No. of fruits /plant
T1	42.00	3.63	21.16	0.81	12.90	18.40
T2	46.43	2.06	23.26	1.13	13.90	22.00
T3	43.80	2.63	23.46	1.16	15.00	20.00
T4	41.46	3.76	24.13	1.20	15.63	25.86
T5	44.16	3.70	23.23	1.11	14.30	22.00
T6	42.50	3.36	17.36	1.06	12.86	16.67
T7	43.23	2.60	16.00	0.95	12.90	15.90
T8	48.00	2.00	14.80	0.81	11.90	12.00
Level of sig.	**	**	**	**	**	**
LSD(0.05)	0.85	0.31	1.34	0.12	1.20	1.10
CV(%)	1.10	5.90	3.60	7.00	4.80	5.00

Table 1. (Cont'd)

Treatments	Fruit length(cm)	Diameter of fruit(cm)	No. of seeds/fruit	No. of ridges/fruit	Fruit yield/plant(g)	Fruit yield(t/ha)
T1	14.36	1.34	21.03	5.80	237.36	8.07
T2	13.93	1.42	22.56	5.70	305.80	10.39
T3	15.23	1.51	21.83	5.35	300.00	10.20
T4	15.16	1.54	23.73	5.73	404.19	13.74
T5	16.25	1.54	29.50	5.60	314.60	10.69
T6	15.56	1.50	21.63	5.50	214.38	7.28
T7	14.50	1.43	19.00	5.60	205.11	6.97
T8	12.90	1.40	16.70	5.55	142.80	4.85
Level of sig.	**	ns	**	ns	**	**
LSD(0.05)	0.70	0.76	1.52	0.58	23.38	0.93
CV(%)	3.00	3.31	4.10	4.00	7.20	8.90

** Significant at 1% level, * Significant at 5% level, ns - Non significant

Here,
T_1= Recommended amount of chemical fertilizers (RD)
T_2= Vermicompost 1.5t/ha + ½ RD
T_3= Organic waste compost 3t/ha + ½ RD Chemical fertilizer
T_4= Vermicompost 1.5t/ha + 2/3rd RD
T_5= Organic waste compost 3t/ha+ 2/3rd RD
T_6= Vermicompost 1.5 t/ha
T_7= Organic waste compost 3t/ha
T_8=Control

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Simulation of different Key factors during composting with non-uniform forced aeration

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ABSTRACT

Composting is widely practiced as a biological treatment method for rapidly decomposable organic solid waste. Among all methods the forced aeration process is very effective with regard to duration of the process and degradation. In this process the air supplies oxygen to the microorganisms for proper biological degradation of organic wastes, and remove excess heat in order to maintain the optimum temperature. The aeration rate is a key process control parameter for the forced aeration composting process for temperature and moisture. Usually a non-uniform aeration is applied in to practical plants. To apply such non-uniform forced aeration, it is necessary to know their effect on different key factors during composting. The effects can be simulated within a very short time by using a mathematical model compared to any experimentation. In this study a developed mathematical model for forced aeration composting was applied to simulate the effect of non-uniform aeration on the different physico-chemical parameters during composting. The results show that: (i) the effects of non-uniform aeration flow rates and the other initial factors on the BVS degradation could be easily simulated by the developed mathematical model for composting mass in a closed system; and (ii) the model could be applied successfully to simulate the effect of initial moisture content on the final degradation of the composting mass.

INTRODUCTION

Municipal solid waste generate in Asian countries contain high amount of organic matter which are extremely decomposable. The decomposable part of organic solid waste emits bad smell to the environment if they are not managed properly. Aerobic composting is widely applied biological method to treat biodegradable organic fraction of municipal solid waste generate. In Europe, the entire organic portion of municipal solid waste is often composted or fermented. Currently many large cities in Asia are planning to erect or improve existing municipal waste composting plants (Bari 2011). Sewage sludge composting has also become very common since the 1970s in the USA (Miller 1991). Different composting technologies depending on the economy of the country, availability of the land, quality of the initial substrate, processing time and process control have been practiced. The main technologies are forced aeration, mechanical turnover in a reactor or in a windrow composting pile. The reactors could be static or slowly rotating and the windrow could be formed in an open field or inside a shelter. Furthermore the process could be batch or continuous; however, the batch process is normally applied for large-scale composting (Sikora et al. 1981, Epstein et al. 1983, Benedict et al. 1986).

The air supplies oxygen to the microorganisms for proper biological degradation of organic wastes, and remove excess heat generated by the microbial activity in order to maintain the optimum temperature in the forced aeration composting process. Insufficient aeration leads to the commencement of anaerobic conditions. The amount of air needs to remove excess heat and moisture from composting mass is typically ten or more times greater than those for biological decomposition. So that when these needs to control the temperature and moisture are met, biological oxygen demands for microorganisms also will be safely satisfied. Usually at the premature stage of composting excess heat should be removed, to maintain a temperature below 60 to 65 ^{o}C, via high rate of aeration. On the other hand, at the later maturation stage a low aeration rate is needed to maintain the aerobic process and at the same time keeping the composting mass warm enough for thermophilic microorganisms and effective pathogens destruction. Therefore, the selection of proper aeration rate is very important to control temperature and moisture during forced aeration composting process. The effect of aeration rate and initial moisture content on composting can be find out experimentally only after a long process time. However, these can be easily simulated using a developed mathematical model as the understanding of the process of composting has significantly advanced. The effect of uniform aeration on forced aeration composting has been presented elsewhere (Bari et. al. 2012). In this paper an attempt is taken to simulate the effect of non uniform or stepped aeration and initial moisture content on the degradation of organic matter using a developed mathematical model as described in the following sections.

INTRODUCTION TO THE MATHEMATICAL MODEL

The basic mathematical model is formulated as a heat balance across the composting mass in a given layer in the reactor as shown in equation (1) and the details are presented elsewhere (Bari and Koenig 2012). This formulation builds upon the findings of previous experimental studies on the heat balance for the pilot scale composting process, self-heating test and kinetic analyses on degradation of biodegradable volatile solids (BVS) under forced aeration composting (Bari and Koenig 2000, Bari et al. 2000, Koenig and Bari 2000).

$$mc_{t,Ln}.c_{pc}\frac{\left(T_t-T_{t-1}\right)_{Ln}}{dt}=\dot{m}a_{i,Ln-1}.c_{pa}.T_{i,Ln-1}-\dot{m}a_{o,Ln}.c_{pa}.T_{t,Ln}+\frac{dBVS_{Ln}}{dt}.H_l-k_c.c_{pc}.mc_{t,Ln}\left(T_{ct,\ln}-T_a\right)-\frac{dw_{Ln}}{dt}.L_e-\frac{dw_{v,Ln-1}}{dt}h_g$$

in kJ/hr (1)

The heat balance in word:
Change of heat energy in the composting mass = Heat inflow through incoming dry air + Heat outflow through outgoing dry air + Biological heat generation by degradation of BVS + Loss of sensible heat to surroundings + Loss of heat due to evaporation + Loss of heat due to change of enthalpy of saturated vapor (in kJ/hr).

The assumptions for the development of the model and solution process have been described in detail elsewhere (Bari and Koenig 2012). There are ten supplementary equations namely: (i) The overall stoichiometric equation of waste degradation, (ii) To calculate mass of dry air inflow into the composting reactor, (iii) To calculate mass of dry air outflow from the composting reactor/layer, (iv) To calculate the remaining *BVS* after time *t,* which is a first order reaction rate equation developed by Bari et al. (2000), (v) To calculate the remaining compost mass after any time *t,* (vi) To calculate the remaining moisture in the different layers of composting mass after any time *t,* (vii) The enthalpy of evaporation of water from different layers of the composting mass L_e (in kJ/kg) at variable temperature, (viii) The enthalpy of saturated water vapor h_g (in kJ/kg) at variable temperature (ix) The dependency of the reaction rate (degradation rate) of *BVS* on temperature, and (x) The specific heat capacity of wet compost, c_{pc}, as depends on the composition of the compost mass.
The supplementary equations mentioned above provide the necessary input information for model equation (1). Nomenclature is given in the following section of the paper.

Nomenclature:
mc = composting mass (waste mixture) in reactor, with *mc* = *FS*+*NVS*+*BVS*+H_2O, kg
mc_t = composting mass in reactor after any time *t*, kg
TS = total solids in the composting mass, with *TS* = *FS* + *VS*, kg
FS = fixed solids (inert mineral matter in the composting mass), kg
VS = volatile solids (organic matter in the composting mass), with *VS* = *NVS*+*BVS*, kg
NVS = non-biodegradable volatile solids, kg
BVS = biodegradable volatile solids, kg
ΔBVS = *BVS* degradation, %
H_2O = water content of the composting mass, kg
T = temperature, ^oC
T_c = initial temperature of composting mass, ^oC
T_{ct} = temperature of composting mass after time *t*, ^oC
T_t = temperature of outlet air or compost mass after time *t*, ^oC
T_a = ambient temperature, ^oC
RH = relative humidity, %
k_T = reaction rate at any temperature *T*, hr^{-1}
k_{25} = reaction rate at temperature 25 ^oC, hr^{-1}
f_w = factor used to estimate the amount of water produced per unit *BVS* degradation, kg/kg
f_{ea} = factor used to estimate the amount of gas produced per unit *BVS* degradation, kg/kg
k_w = factor used to estimate the reaction rate at

$\dot{m}a_i$ = dry air mass inflow into the compost reactor, kg/hr
$\dot{m}a_o$ = dry air mass outflow from the compost reactor, kg /hr
Q_i = airflow rate in m^3/m^2.hr
O_{2t} = oxygen concentration of the air at any time t, %
w = moisture content of composting mass, % of wet mass
w_v = mass of water vapor (saturated) in dry air, kg/kg
w_{vi} = mass of water vapor in dry inflow air, kg/kg
w_{vo} = mass of water vapor (saturated) in dry outflow air, kg/kg
dt = one hour time interval
t = time, hr, = 1, 2, 3 …… t
moisture contents below 45%
c_{pc} = specific heat capacity of wet composting material, kJ/kg.ºC
c_{pa} = specific heat capacity of dry air, kJ/kg.ºC
H_l = heat energy generated by the degradation of BVS, kJ/kg
h_g = enthalpy of saturated water vapor, kJ/kg
L_e = latent heat of evaporation of water, kJ/kg
k_c = specific heat transfer coefficient, h^{-1}, with $k_c = (U.A)/(mc.c_{pc})$, where
U = overall coefficient of heat transfer through top and side of the filled reactor, kJ/h.m^2.ºC,
A = total surface of top and side of filled reactor, m^2
L = layer of composting mass
n = layer nos. starting from bottom = 1, 2, …….6

METHODOLOGY

At the development stage of the mathematical model, uniform aeration is considered in the previous study (Bari and Koenig 2012) as mention in the section 1. In fact, the aeration is usually varied according to practical need throughout the composting period. In other word, non-uniform aeration is applied in the composting plants. Usually at the early stage of composting a high rate of aeration is applied to remove excess heat, to maintain a temperature below 60 to 65 ºC in the composting mass. On the other hand, at the later stage (stabilization stage) a low aeration rate is needed to maintain the aerobic biological degradation process and at the same time keeping the composting mass warm enough for thermophilic microorganisms and effective pathogen destruction. In this study aeration with three steps (or is called non-uniform aeration) is considered for all simulations. Total duration is divided in three parts of time and three different sets non-uniform aeration is proposed in this study as indicated by A1, A2 and A3 as shown in Table 1. At the beginning part comparatively higher aeration is applied and subsequently the aeration decrease in the later steps as mention in the 2nd, 3rd and 4th columns of Table 1. Three different parts of duration are 7 days (from 0 to 7 days), 7 days (from 8 to 14 days) and 13 days (from 15 to 28 days). For the proposed aeration set A1, an aeration rate 7.5 m^3/hr is applied for the first 7 days, then the aeration rate 4.5 m^3/hr is applied for 8 to 14 days and finally an aeration rate 3.0 m^3/hr is applied for last 13 to 28 days.

The initial waste properties were selected (on the basis of experimental results, Bari 2011) as initial wet weight of waste mixture mc = 750 kg/m^2, bulk density = 550 kg/m^3, initial height = 140 cm, initial moisture content w = 60.0%, initial fixed solids FS = 4%, initial non-biodegradable volatile solids NVS = 48%, and initial biodegradable volatile solids BVS = 48%. Very clear changes among temperature curves due to application of non-uniform aeration for three simulations are shown in Figures 1a, 2a and 3a which are again related to the degradation pattern. Usually unidirectional aeration distributes the temperature of different layers with low temperature near inlet and high temperature near the outlet of the air flow.

RESULTS AND DISCUSSION

Effect of Non-uniform Aeration

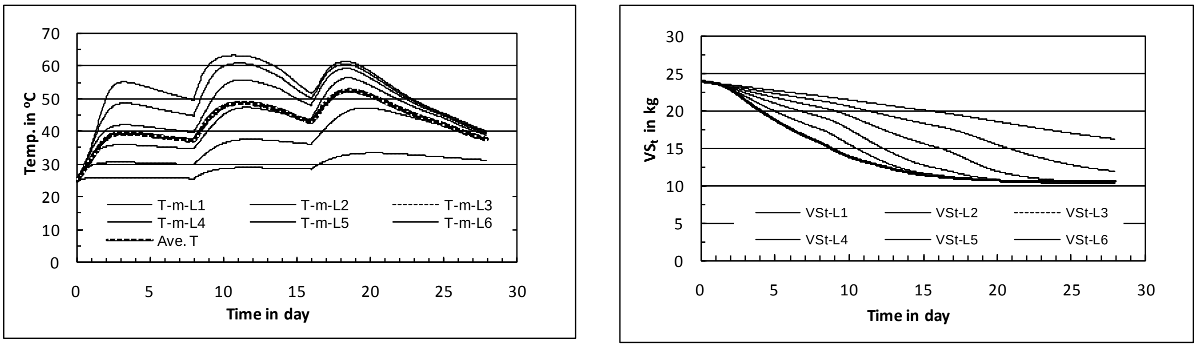
Very clear changes among temperature curves due to application of non-uniform aeration for three simulations are shown in Figures 1a, 2a and 3a which are again related to the degradation pattern. Usually unidirectional aeration distributes the temperature of different layers with low temperature near inlet and high temperature near the outlet of the air flow.

The effects of selected three sets of non-uniform aeration on temperature and on other key parameters such volatile solids, oxygen concentration and moisture content during composting in a closed system are shown in Figures 1, 2, and 3. High degradation of biodegradable volatile solids occurs in simulation A2 considering medium non-uniform aeration as shown in Figure 2b as compared to the curves shown in Figures 1b, and 3b. Total air used in three simulations is 3173, 2115 and 1058 m^3, respectively, as presented in Table1. Degradation of biodegradable volatile solids (*ΔBVS*) for A1, A2 and A3 are 51.5%, 54.4% and 33.7%, respectively. It means more aeration do not work always.

Table 1 Three sets of non-uniform aeration, the amount of air used and percentage BVS degradation simulated composting process

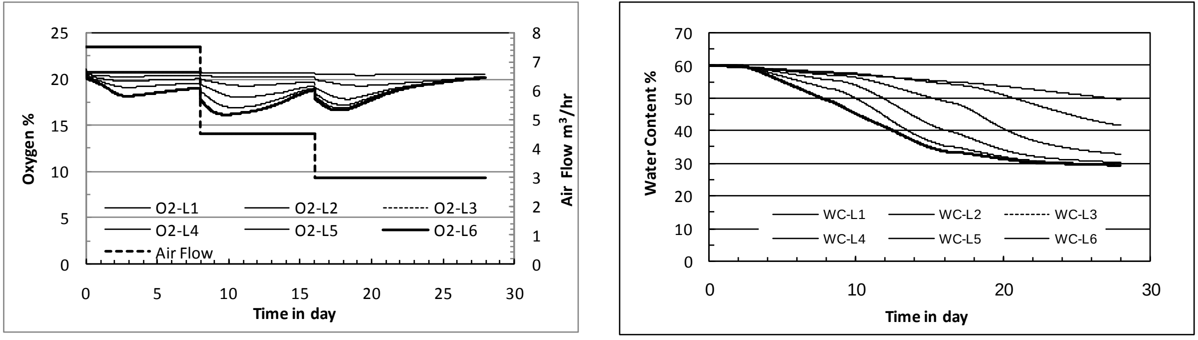
Simulation	Air flow rate in three parts of total duration, m^3/hr			Total air used,	BVS degradation,
	0 -7 d	8 – 14 d	15 – 28 d	m^3	%
A1	7.5	4.5	3.0	3173	51.5
A2	5.0	3.0	2.0	2115	54.4
A3	2.5	1.5	1.0	1058	33.7

The changes in oxygen consumption are also very clear as shown in Figure 1c, 2c and 3c. At simulation A1 with higher step aeration the oxygen concentration always remain above 15% throughout the composting period and in A2 the concentration sometimes falls below 15%. On the other-hand the O_2 concentration remains very near to 15% and at the beginning it falls to 7 to 8%. The changes in moisture content are shown in Figure 1d, 2d and 3d. Due to low aeration and consequently attainment of higher temperature a low *BVS* degradation occur in simulation A3 as shown in Figure 3b. At higher ceiling temperature the biological reaction rate becomes almost zero as presented elsewhere (Bari and Koenig, 2007).



a. Change in temperature

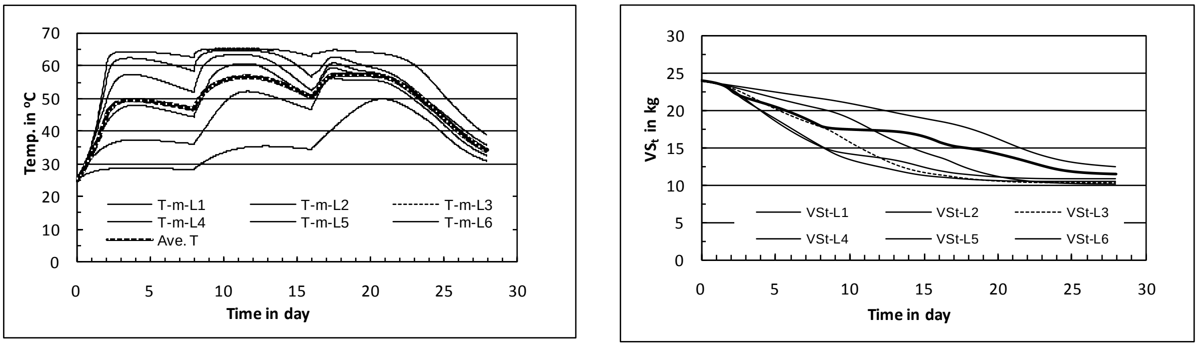
b. Change in BVS



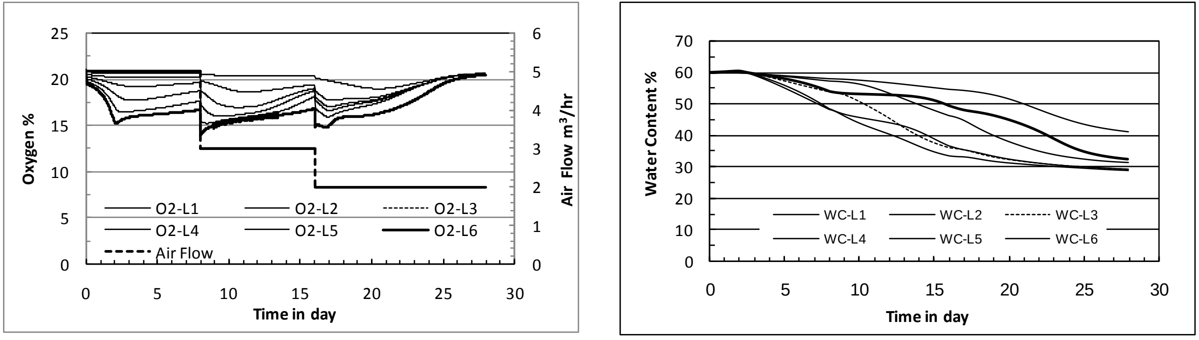
c. Change in oxygen concentration

d. Change in water content

Figure 1. Effect of non-uniform aeration on different factors, simulation A1

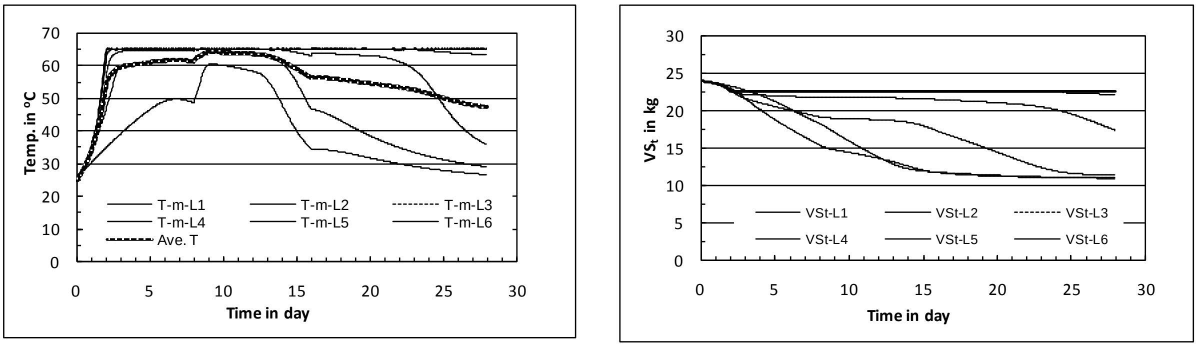


a. Change in temperature b. Change in BVS

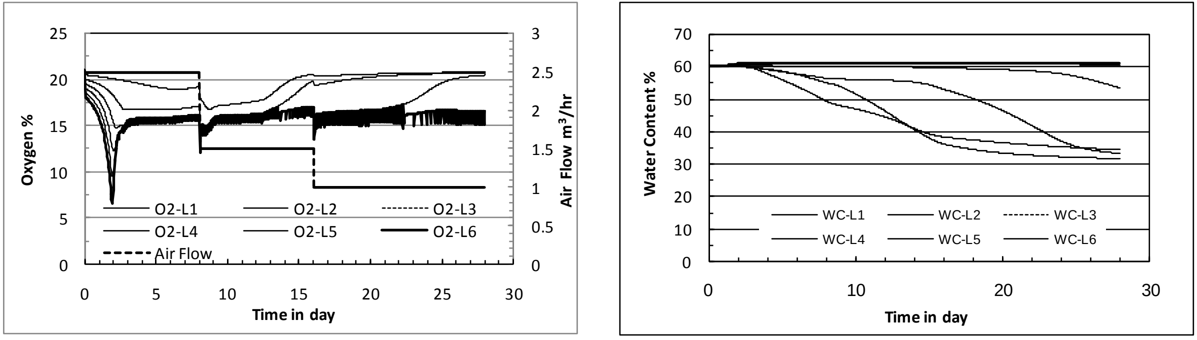


c. Change in oxygen concentration d. Change in water content

Figure 2. Effect of non-uniform aeration on different factors, simulation A2.



a. Change in temperature b. Change in BVS

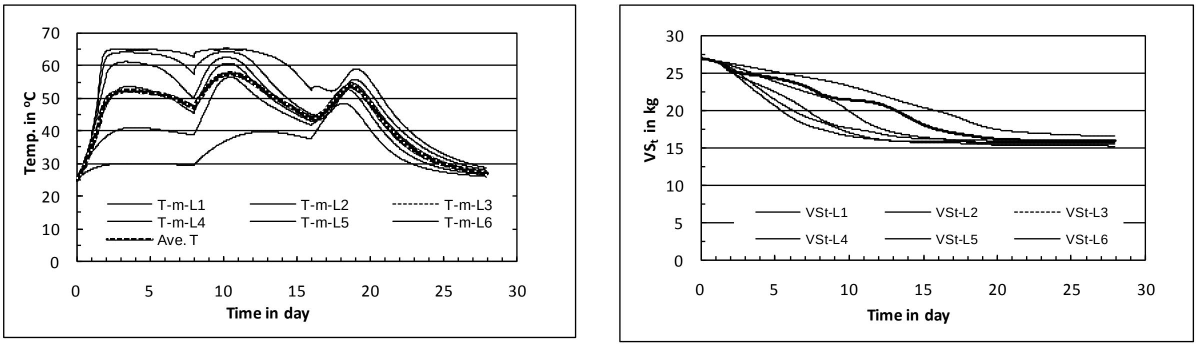


c. Change in oxygen concentration d. Change in water content

Figure 3. Effect of non-uniform aeration on different factors, simulation A3

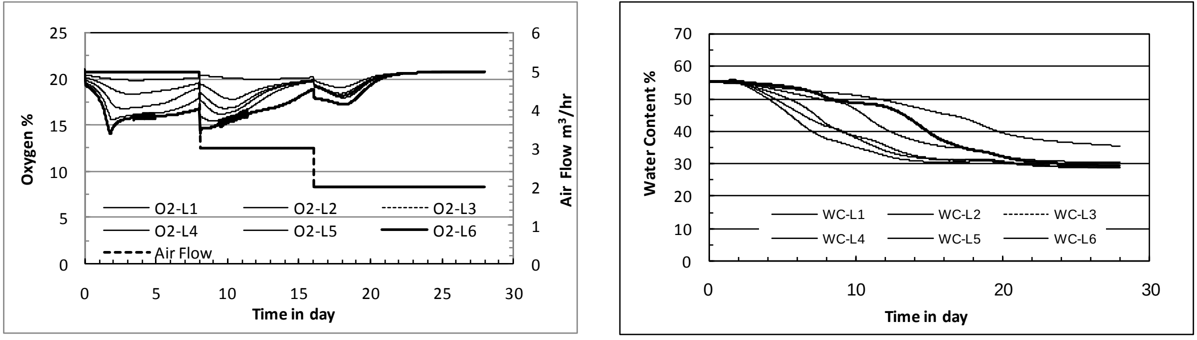
Effect of Initial Moisture Content with Non-uniform Aeration

For this particular analysis, upflow non-uniform aeration of type A2 as shown in Table 1 is used for the composting. The selected initial moisture content are *w* = 60%, 50% and 40%. The initial waste properties were selected as initial wet weight of waste mixture *mc* = 750 kg/m^2, bulk density = 550 kg/m^3, initial height = 140 cm, *FS* = 4%, *NVS* = 48%, and *BVS* = 48%. Figures 2, 4, 5 and 6 present the patterns of simulated temperature, pattern of *BVS* degradation, changes in percentage oxygen in the waste air, and changes in moisture content during the composting process for different initial moisture content as 60% 55%, 50% and 40%, respectively. Degradation of biodegradable volatile solids (*ΔBVS*) occurs high in simulation considering 60% initial moisture content as shown in Figures 2b, 4b, 5b, and 6b. Extent of degradation of *BVS* are 54.4%, 41.4% 29.7% and 20.4%, for moisture content 60%, 55%, 50% and 40% respectively, for simulations A2, M1, M2, and M3 asn indicated in Table 2. Dependency of *ΔBVS* with corresponding temperature and oxygen consumption can be clearly identified from the series of the Figures2, 4, 5 and 6. Changes in various *BVS* for different moisture content are compared with corresponding Figures 2a, 4a, 5a and 6a for temperature. Tendency of low rise in temperature due to low initial moisture content in the composting mass and consequently low *BVS* degradation are serially shown in Figures 2a, 4a, 5a and 6a. Similarly tendency of low consumption of oxygen due to low moisture content and consequently low *BVS* degradation are serially shown in Figures 2c, 4c, 5c and 6c.



a. Change in temperature

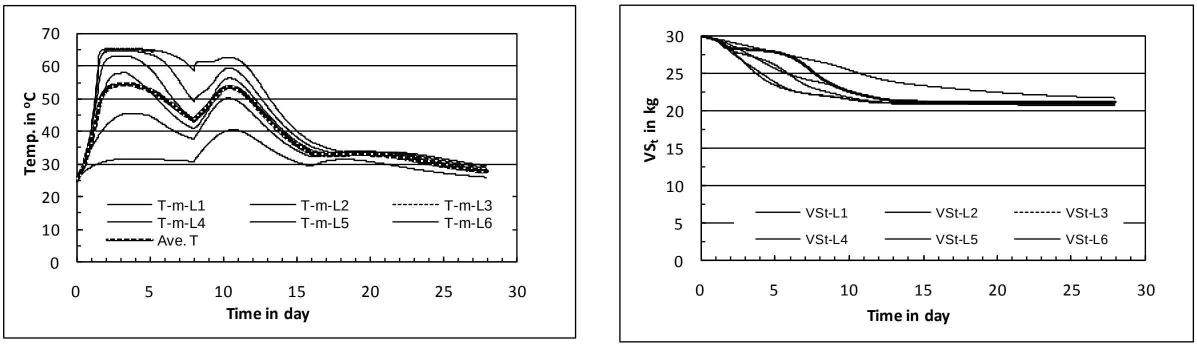
b. Change in BVS



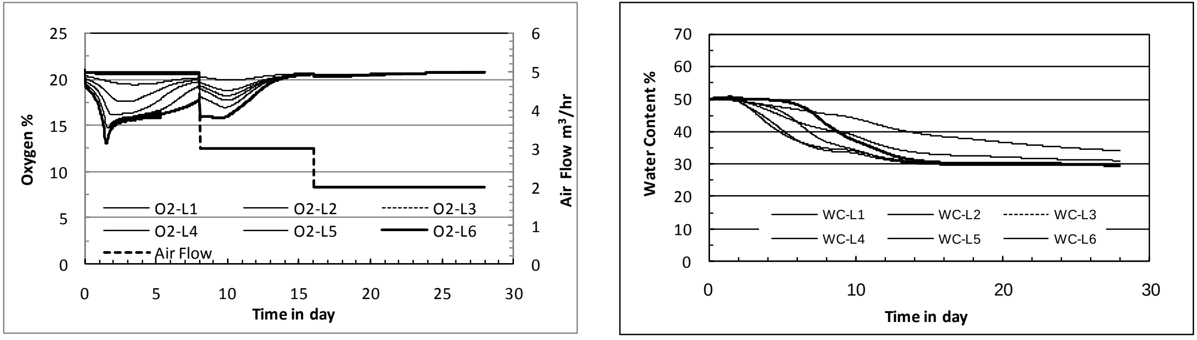
c. Change in oxygen concentration

d. Change in water content

Figure 4 Changes on different composting factors due to initial moisture content 55%.

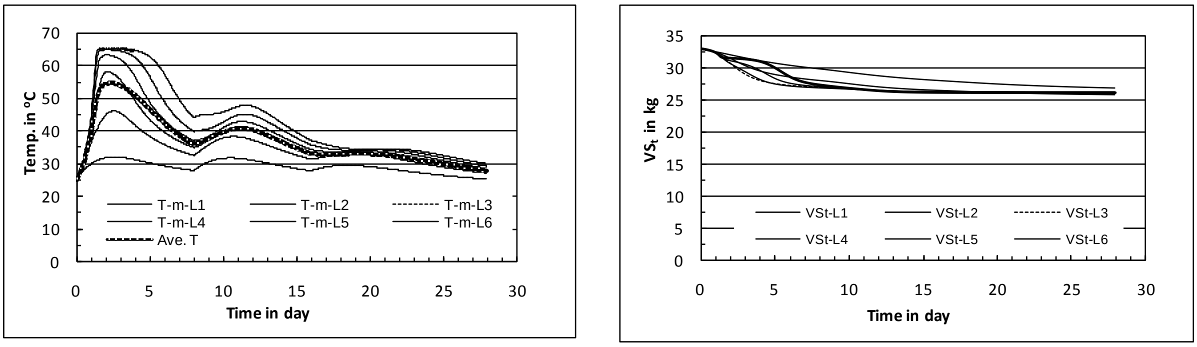


a. Change in temperature b. Change in BVS

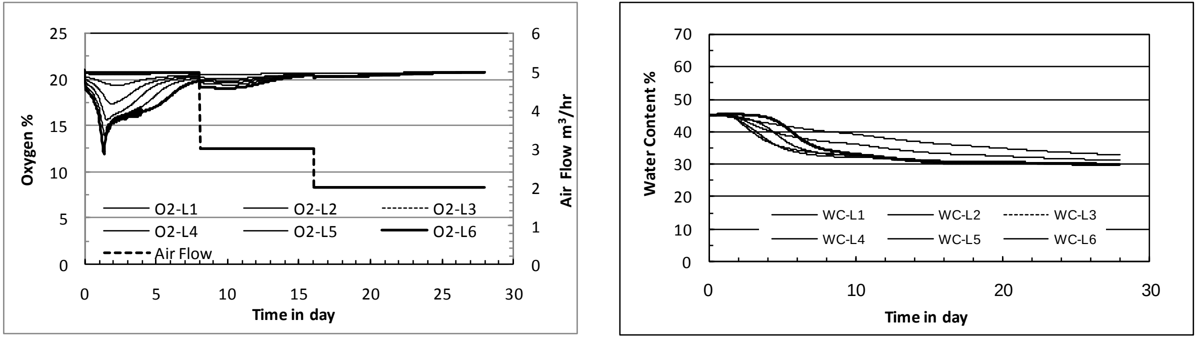


c. Change in oxygen concentration d. Change in water content

Figure 5 Changes on different composting factors due to initial moisture content 50%.



a. Change in temperature b. Change in BVS



c. Change in oxygen concentration d. Change in water content

Figure 6 Changes on different composting factors due to initial moisture content 45%.

Table 2 Percentage BVS degradation for different initial moisture content a single non-uniform aeration composting simulation of A2

Simulation	Initial Moisture Content, %	BVS degradation, %
A1	60	54.4
M1	55	41.4
M2	50	29.7
M3	45	20.4

ENGINEERING SIGNIFICANCE

Successful application of a developed mathematical model are presented in this paper using only using all important input parameters such as the properties of initial waste mixture, airflow rates, and ambient conditions. It has predicted the instantaneous status of different physico-chemical parameters including temperature oxygen consumption and *BVS* degradation. Its versatile application shows that, the model is also very flexible and allows changes in the input parameters at any time. Being based on solid scientific fundamentals, it can be used for conceptual process design, studies on the effect of ambient conditions, optimization studies in existing plants, and process control. The model is very effective in simulating the composting process for varying engineering requirements and instantly predicting the output parameters.

CONCLUSIONS

- The effects of non-uniform aeration rates and the initial moisture content on the BVS degradation could be easily simulated by the developed mathematical models for multi layer composting mass in a closed system such as tunnel composters.
- Tendency of low rise in temperature due to low initial moisture content in the composting mass and consequently low *BVS* degradation are observed in different simulations.
- Similarly tendency of low consumption of oxygen due to low moisture content and consequently low *BVS* degradation are also clearly observed.

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