3 LANDFILL STUDIES

3.1 Introduction

Landfill is the most preferred method of municipal solid waste (MSW) disposal in Malaysia. As the fastest developing state in Malaysia, Selangor generated approximately 3,833 tonnes of MSW in 2008 of which approximately 95% were disposed off into landfills. The need for fast improvements in the waste management and disposal method is very crucial to prevent detrimental effects to the environment. Until 2004, most of the MSW collected by the municipalities and waste collectors were sent to 10 landfills throughout Selangor, out of which nine were open dumps or controlled dumps. This chapter collates data from the nine non-sanitary landfills in Selangor. These include waste composition, waste generation, landfill leachate and other issues pertaining to municipal waste landfill and pollution treatment from the non-sanitary landfills in Selangor. These landfills include Panchang Bedena landfill, Kampung Hang Tuah landfill, Kundang landfill, Sungai Sedu landfill, Ampar Tenang landfill, Bukit Beruntung landfill, Kerling landfill, Hulu Yam Bharu landfills and Sungai Kembong landfill. The first four landfills were handed over to Alam Flora (P) Ltd. (AFSB) by the local municipality in 1995. Bukit Beruntung landfill, Kerling landfill and Hulu Yam Bharu landfills are under The District Council of Hulu Selangor while Sungai Kembong landfill was under the jurisdiction of Kajang Town Council.

The term ‘landfill’ used in this chapter may not apply to its standard definition since most of the MSW disposal sites in Malaysia are open dumps, which lack proper lining system as well as other amenities. However, they are referred to as ‘landfills’ which are classified into four categories Class I, Class II, Class III and Class IV.

Air Hitam Sanitary Landfill was the only sanitary landfill then and it catered for the disposal of waste from Kuala Lumpur till 2006. The closing of Air Hitam Landfill in Puchong had lead to the operation of two new sanitary landfills namely Bukit Tagar Sanitary Landfill and Jeram Sanitary Landfill. The former which is under the management of KUB Berjaya Berhad and the Ministry of Housing and Local Government (MHLG) is meant to dispose waste collected from Kuala Lumpur. The latter is to accommodate the disposal of waste from Selangor. Jeram Sanitary Landfill whose jurisdiction is held by the Selangor State Government is managed by World Wide Landfill (P) Ltd..

These sanitary landfills are well planned and engineered to prevent the risk of environmental contaminations. The technologies applied ensure that pollutants generated by these landfills will not contaminate the environment as long as the monitoring and preventive actions are taken care off. On the other hand, the remaining landfills pose threat to the environment due to the lack of appropriate measures to curb pollution.

Descriptions of the nine landfills are summarized in Table 3.1 and detailed in the consecutive paragraphs.

Table 3.1: Summary of landfills under study.

No	Landfill name and operation	Tonnage of waste received daily	Location and area size of landfill	Service area	Landuse prior to landfill
1	Panchang Bedena landfill (1989)	60 tonnes (active)	Sabak Bernam district (15 acres)	Sabak Bernam, Tanjung Karang, Sekinchan, Bagan Terap and Sungai Besar	palm-oil plantation
2	Kampung Hang Tuah (KHT) landfill (1989)	100 tonnes (active)	Kuala Selangor district (20 acres)	Batang Berjuntai, Batu Arang, Kuala Selangor, Puncak Alam and Sungai Buloh-Kuala Selangor route.	Swamp area
3	Kundang landfill (1996)	400 tonnes (closed in 2007)	Rawang district (50 acres)	Selayang municipality, Kuala Lumpur and Rawang	Tin- mining
4	Sungai Sedu landfill (1992)	200 tonnes (active)	Banting district	Telok Panglima Garang, Banting and Jenjarom	Sand mining
5	Ampar Tenang landfill (1995)	100 tonnes (active)	Sepang district (10 acres)	Sepang, Salak Tinggi, Nilai and Dengkil	Palm-oil plantation
6	Bukit Beruntung landfill (1992)	80 tonnes (active)	Hulu Selangor district (5 acres)	Serendah, Bukit Beruntung and Bukit Beruntung industrial area	Secondary forest area
7	Kerling landfill (1993)	150 tonnes (closed in 2006)	Selangor-Perak boundary (15 acres)	Kerling, Lembah Beringin, and Ulu Bernam	Secondary forest
8	Hulu Yam Bharu landfill (1990)	50 tonnes (closed in 2008)	Hulu Yam Bharu (6 acres)	Kuala Kubu Bharu, Hulu Yam Bharu and Hulu Yam Lama	Tin-mining
9	Sungai Kembong (1989)	600 tonnes (closed in 2009)	Hulu Langat district (80 acres)	Semenyih, Kajang and Bangi.	Secondary forest

Note: All landfills are active during the study period (2002-2006).

Panchang Bedena landfill which is servicing a total population of approximately 31,000 people was selected due to the presence of clay layer, which made it suitable for construction of eight waste cells (Plate 3.1). The landfill is undergoing upgrading since its handover to AFSB in 2004.

Plate 3.1: Active cells and waste disposed at Panchang Bedena

Kampung Hang Tuah (KHT) landfill received approximately 70 tonnes/day. The economic activities include agricultures and industries and this resulted in a very mixed waste composition (Plate 3.2). The landfill has been upgraded by AFSB with the installation of fence, gas pipe system and others (Plate 3.3). Recovered materials such as metal, paper and others were recycled by scavengers (Plate 3.4).

Plate 3.2: The disposal of highly mixed wastes in KHT landfill.

Plate 3.3: Gas pipes at Kampung Hang Tuah landfill

Plate 3.4: Material recovery by the scavengers from the KHT landfill’s waste stream.

Kundang landfill was taken over by AFSB in 1998 and it was receiving approximately 400 tonnes of waste daily (Plate 3.5 and Plate 3.6). This landfill has the capacity to operate until 2008, but due to leachate over-flow into the adjacent river, it was ordered to cease its operation in February 2006 by the federal government. The landfill is currently closed and under the post-closure procedure.

Plate 3.5: Recovered materials collected from waste disposed in Kundang landfill during its operation.

Plate 3.6: Scavenging activities in Kundang landfill prior to its closure in February 2006.

Sungai Sedu landfill receives approximately 200 tonnes of waste daily (Plate 3.7). The main activities in this area are industrial, commercial and agriculture. This landfill is currently upgraded to Class IV.

Plate 3.7: Some industrial waste disposed at Sungai Sedu landfill

Ampar Tenang landfill receives more than 100 tonnes of waste daily (Plate 3.8). This landfill which was a mere open dumps is being upgraded with the installation of gas piping system, leachate collection and treatment pond, fencing, daily waste cover and other amenities to a Class IV standard.

Plate 3.8: Waste cell and gas pipe at Ampar Tenang landfill.

Bukit Beruntung landfill, which is visible from the North-South Highway (PLUS), receives approximately 80 tonnes of waste daily. The depression in the hilly area allows the expansion of the disposal site. Lack of enforcement by local municipality resulted in illegal dumping of MSW along the road to the landfill site (Plate 3.9). The landfill lacks any facilities that it falls under Class I. Scavenging activities is very active in the landfill where recyclable materials are recovered (Plate 3.10).

Plate 3.9: Waste disposed at Bukit Beruntung landfill and the illegal dumping along the way to Bukit Beruntung landfill

Plate 3.10: Recovered materials from Bukit Beruntung landfill.

Kerling landfill received approximately 150 tonnes of waste daily (Plate 3.11), which was pushed down into the ravine. Though it has the capacity to operate for more than 10 years, the landfill was closed to provide way for North-South bound railroad (Plate 3.12). Similar to Bukit Beruntung, Kerling was a mere open dumping site for MSW. Since it is impossible to retrieve or recover materials from this landfill, it was chosen to dispose sensitive materials such as police uniform, boots and others.

Plate 3.11: Wastes were dumped and pushed into the ravine at Kerling landfill.

Plate 3.12: Closing of Kerling landfill for the railroad construction.

Hulu Yam Bharu landfill received approximately 50 tonnes of waste daily (Plate 3.13). It also operates as an open-dump due to the absence of landfill liner, fence, piping system and others. The landfill also becomes an attraction to scavengers (Plate 3.14).

Plate 3.13: Waste disposal at Hulu Yam Bharu landfill

Plate 3.14: Scavenging activities by human and animals at Hulu Yam Bharu landfill.

Sungai Kembong landfill receives 600 tonnes of waste daily (Plates 3.15). The landfill received various industrial wastes and was also very active with scavenging activities in recovering materials for recycling (Plate 3.16).

Plate 3.15: Active cells releasing leachate into the pond

Plate 3.16: Scavenging activities in Sungai Kembong.

These landfills require major attention in order to improve the current waste management system. Upgrading the landfill and the establishment of an effective waste management is very crucial to produce an efficient waste disposal system for the state in particular. Furthermore, it could become a guideline of waste management improvement for other states as a whole. Therefore, background information and data should be compiled from all of these landfills to identify the improvement options. The consecutive sections discuss the materials and methods in compiling data, analysis of data, experiment on various parameters and the results and discussion of the landfill studies.

3.2 Materials and Methods

The study was conducted in three phases: first phase covered all nine landfills (operating non-sanitary landfills in Selangor), second phase covered three selected landfills representing urban, sub-urban and rural landfills, and detail studies in the final phase involved only one landfill. Studies include waste characterization and composition, waste generation, waste analysis, leachate analysis and leachate treatment. Subsequent paragraphs detail the method of each section.

3.2.1 Landfill Studies

3.2.1.1 Waste Characterization/ Composition Studies at Disposal Sites

Stratified sampling was employed to allow non-bias results (Terashima *et al*., 1984) where source of waste namely residential, industrial, institutional or commercial become the main stratum of selection at each landfill. Each day, five to seven garbage trucks and compactors were selected randomly according to their stratum. Wastes from these lorries were dumped onto a tarpaulin at a designated area within the landfill compound. The amount of waste from these lorries ranged from 10 to 15 tonnes. Quartering methods were applied where each pile of waste was divided into four sections. Two quarters will be retained while the remaining two were rejected. The selected quarters were quartered again where half of the section was selected to achieve approximately 100 to 250 kg of waste per section. The quartering method applied in the study allowed a more random waste sampling since waste collections by the lorries were conducted based on designed routes. The selected section was sorted into nine groups, namely organic waste, plastics, paper, wood, textile, rubber, metal, glass and miscellaneous (Plate 3.17). The separated wastes were weighed and the volume determined. Studies were conducted for a week in all landfills, sampling domestic wastes, institutional waste, commercial waste, and non-hazardous industrial waste. The data was computed and analyzed to determine the waste composition in terms of percentage, density and others. The results obtained from the studies and the socio-economic levels are used to identify the most suitable landfills to represent urban landfill, rural landfill and sub-urban landfill. The studies were conducted from November 2002 to November 2003.

Plate 3.17: Waste sorting at Kundang landfill

Selection of the three landfills was based on the amount of waste received by the landfill, the economic level of the area and the population density of area served by the landfill. Similar to the first phase, the waste analysis were conducted in the landfill from randomly selected lorries. Detailed analysis for phase two of the study included the separation of waste into 28 groups. The detail separation will allow the identification of various options in managing the waste including recycling, combustion and composting. The details analysis required the waste to be sorted according to the waste group as in Table 3.2. Separated wastes were weighed and the volumes were determined to compute the percentage based on fresh weight and the density. The studies were conducted in 2003-2006.

Table 3.2: Detailed composition of MSW.

Waste Types	**Waste Group**	**Example**
Kitchen waste	Food waste	Vegetable, fruits skin, left-over food etc
	Food (not-consumed)	Expired food, rotten food etc.
Paper waste	Mixed paper	Coloured paper, heterogeous papers
	Newsprint	Newspaper
	Phone book	Phonebooks
	Magazine	Magazine, glossy paper
	White paper	Computer paper, good quality papers etc
	Corrugated paper	Box, cartons etc.
Plastic waste	Plastic (rigid)	Plastic toys, plastic pails etc.
	Plastic (film)	Plastic bags and non-rigid, film like plastics
	Plastic (polystyrene)	Food containers, electrical appliances fixing polystyrene etc.
	Disposable diapers	Diapers.
Textile waste	Textile	Clothes, rags etc.
Rubber/ leather	Rubber/ leather	Shoes, tyres, etc.
Wood	Wood	Part of wooden furniture, wooden crates etc.
Garden waste	Garden waste	Leaves, tree branches, grass etc.
Glass	Clear glass	Non- coloured glass, window glass etc
	Coloured glass	Coloured or dark glass etc
Metal	Metal	Plumbing pipes, parts of electrical appliances etc.
	Tin	Food can etc.
	Aluminium can	Drinking cans
	Other aluminium	Aluminium foil etc.
Miscellaneous	Non-metal	Non-metal materials
	Hazardous waste	Batteries, aerosol cans, medicine etc.
	Sand/dirt	Sand, dirt, fine materials etc.
	Other organic	Non-food materials etc.
	Other non-organic	Ceramic, inorganic materials etc.
	Bulky waste	Furniture, electrical appliances etc.

3.2.1.2 Waste Analysis

Waste analysis involved the determination of physical, chemical and biological characteristics. All analysis was conducted according to the USEPA standard methods. Among the analysis conducted include waste moisture content, pH, salinity, and conductivity.

3.2.2 Waste Generation Studies at Generation Points

Waste generation studies were carried out in Kundang (urban), Sungai Sedu (sub-urban) and Sungai Besar (rural) area. The generation studies involved approximately 800 household from each study area. This study focused on waste generation rate at source, and the impacts of income. Since the study was also conducted overlapping fasting month, it was possible to compare. Also, studies were conducted according to the income level of the residents which include high-income groups (above RM5,000), middle-income groups (RM1,000- RM5,000) and low-income groups (below RM1,000) where the ratios are 2:2:1 respectively, based on UNICEF (2004) statistics. Each resident was briefed with background of the project and supplied daily with two sets of garbage bags to separate wet waste such as kitchen waste, and dry waste like paper, plastics and others. The study was conducted for a week during the fasting month and a week during the non-fasting month from 2004 to 2007). The selection of fasting months as one of the parameter studied is based on the hypothesis that more wastes were generated in fasting months than that of non-fasting months (Dusuki, 2008). Randomly selected wastes were sampled for analysis.

3.2.3 Waste Treatment Options

Treatment options attempted for the waste collected include vermi-composting and Refuse-derived Fuel (RDF) conversion. Consecutive paragraphs detail the methodology for the treatment options.

3.2.3.1 Vermi-composting

Vermi-composting was conducted in flower pots to provide information on a small-scale composting which would be suitable for individual household. Clay flower pots were used since clay is a better temperature regulator than others. Flower pots are also easily available and can be placed anywhere in the house. Worms used were obtained locally (Plate 3.18). Vermi-composting of kitchen wastes were set-up accordingly with various additives (Table 3.3).

Plate 3.18: Worms used in the vermi-composting trials.

Table 3.3: Vermi-composting set-ups 1 (VCS1).

Label	Combination	Ratio
KW	Kitchen Waste (KW)	1
KW + GC	KW and Grass Clippings (GC)	2:1
KW + GM	KW and Goat Manure (GM)	2:1
KW + VC	KW and Vermi-compost (VC)	2:1
KW + GS	KW and Garden Soil (GS)	2:1

The set-ups were prepared with sufficient aeration and drainage system for vermi-composting as illustrated in Figure 3.1.

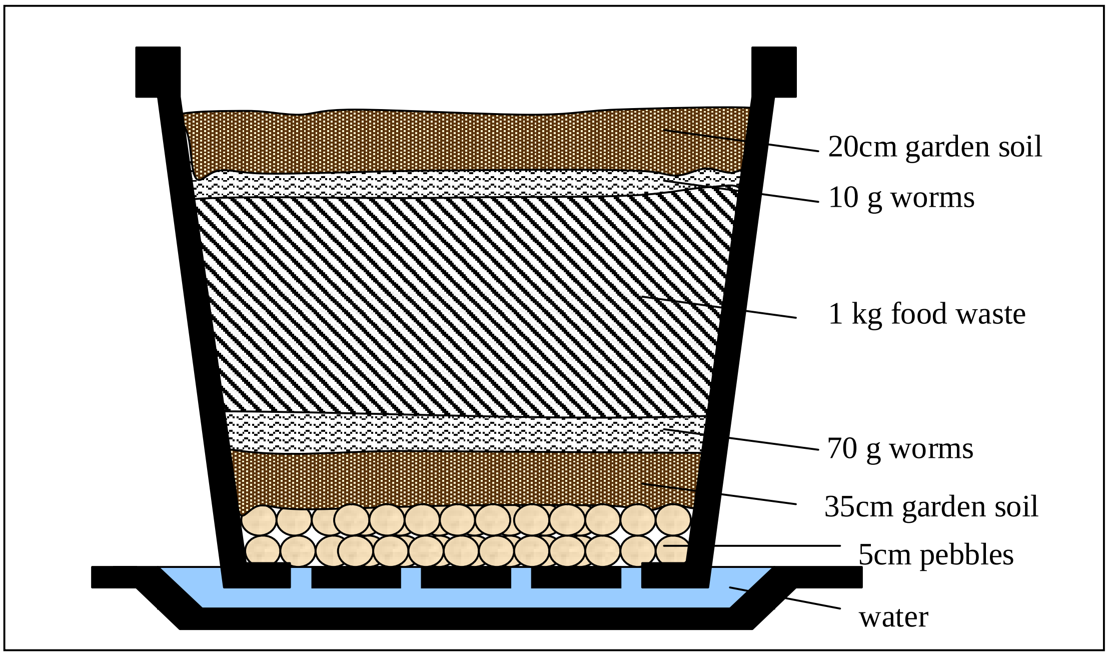


Figure 3.1: Schematic diagram of vermi-composting first setup.

The bottom plate with water was to ensure that worms escaping from the pot will not be dehydrated. The first compost set-ups were allowed to progress for a month before the compost was analyzed for total C, total N, total P, total K, total Mg, and total Ca. Other metal elements were also analyzed to evaluate the possible presence of foreign components in compost from kitchen waste. The analyses were conducted based on the standard methods (Appendix 3.1). Second vermi-composting was set-up similar to the first set-up to determine the possibility of soil enhancement. Food wastes were buried in flower pot with worms and covered (Figure 3.2).

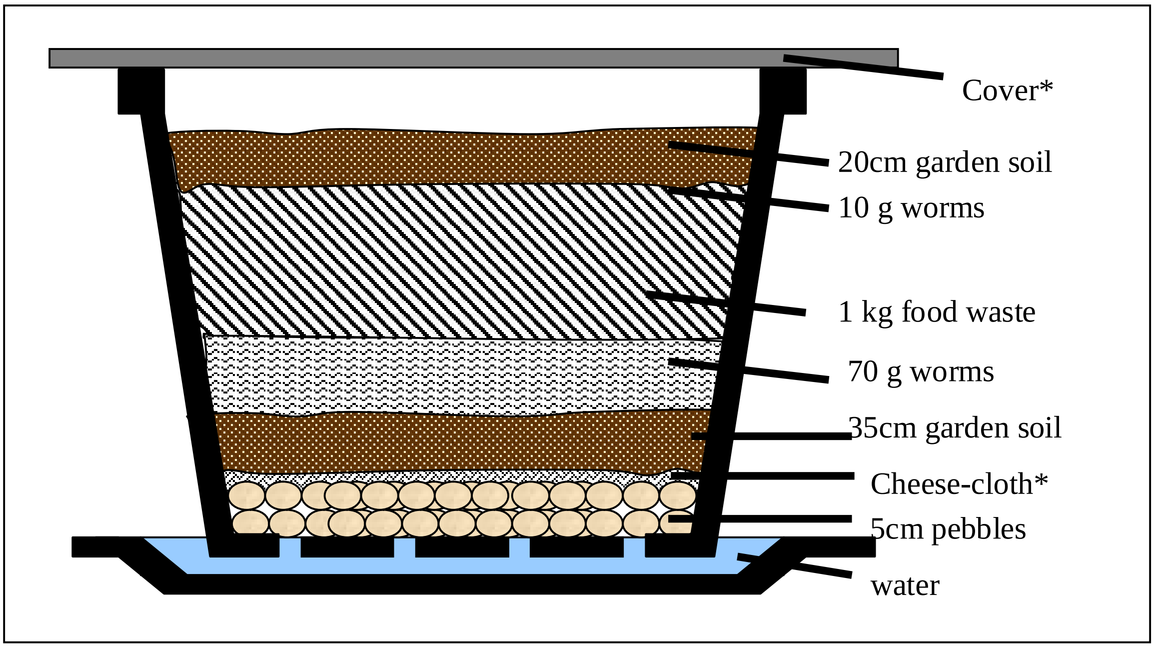


Figure 3.2: Schematic diagram of vermi-composting in the second setup.

The analysis in the second vermi-compost included the total C, N, P, K, Mg and Ca comparing the soil nutrient before and after the introduction of food waste and worms. Throughout the vermi-composting process, moisture in the pots was maintained at 40-60% to prevent dehydration, which will cause death to the worms in the system.

3.2.3.2 RDF conversion

Waste samples suitable for combustion were collected from various sampling and air-dried for 24 hours. Samples were shredded to the appropriate size and palletized (Plate 3.19). The pallets were weighed and analyzed with the bomb calorie-meter (Model C2000) to determine the calorific values (Appendix 3.2).

Plate 3.19: Shredded and palletized samples from various waste types

3.2.4 Leachate Analysis

Leachate samples were collected from the nine landfills using grab method. The analysis conducted included physical, chemical and biological analysis. The procedures involved in sampling and analysis is based on the APHA Standard Methods for Analysis of Water and Wastewater (1998) listed in Appendix 3.1

3.2.5 Leachate Treatment System

Leachate treatment included chemical and biological treatment. Leachate was collected from Kundang landfill during its operation (2006) and after its closure (2007). The leachate underwent chemical and biological treatment and the reduction in pollution parameters were determined to identify the efficiency of the applied system. The physico-chemical treatment involved coagulation and flocculation procedure with ferric chloride and alum at different concentration and different pH. Table 3.4 summarizes the coagulation and flocculation applications in the leachate treatment studies.

Table 3.4: Physico- chemical studies of leachate via coagulation and flocculation process.

Experiment	Alum	Ferric Chloride
Series of concentration applied (g/L) pH at initial stage	0, 0.2, 0.3, 0.5, 0.8, 1.0, 1.2 7.0	0, 0.2, 0.4, 0.5, 0.8 7.0
Concentration (g//L) Series of pH applied	0.5 3.0, 4.0, 5.0, 6.0, 7.0, 8.0	0.8 3.0, 4.0, 5.0, 6.0, 7.0, 8.0
Speed	Rapid: 250 rpm (3 minutes) Slow: 90 rpm (30 minutes	
Settling time	30 minutes	

Leachate analyses were conducted before and after coagulation and flocculation, followed with further leachate treatment with microbial cocktail. Microbes were isolated from landfill and purified to pure cultures. Since the pure cultures of the microbes were not identified, the inoculation into broth is conducted at random. Approximately 35 bacterial cultures were inoculated into each nutrient broth (NB1- NB3) while each potato dextrose broth (PDB1- PDB3) were inoculated with approximately 30 fungal cultures. The broths were incubated at $37^{0}C$ for approximately 30 hours where the optimal density of the microbial cocktails reached 200-250 mg/L. Plate 3.20 depicts the bacterial and fungal cocktails used in the biological treatment of leachate.

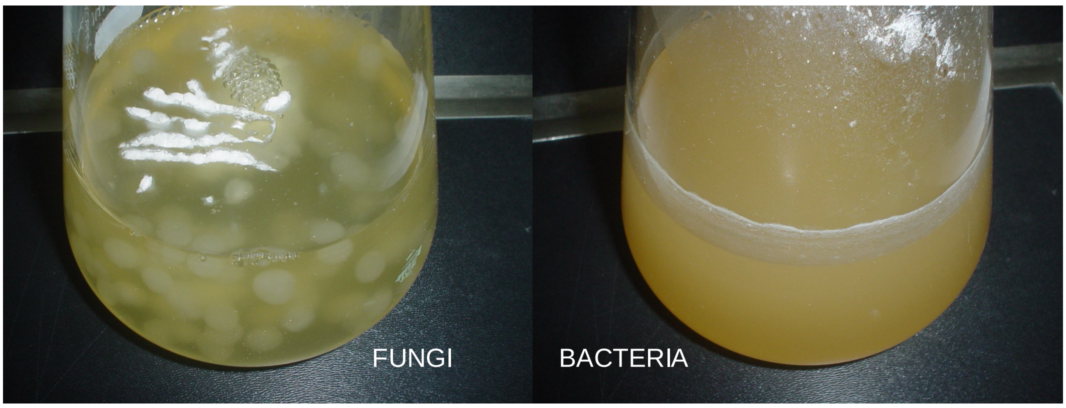


Plate 3.20: Microbial cocktails used in the biological treatment of leachate.

A 100 mL of the microbial cocktail was added into one liter of leachate which was pretreated with 0.3g ferric chloride at pH 5. The experimental set-up is shown and summarized in Figure 3.3 and Table 3.5, respectively.

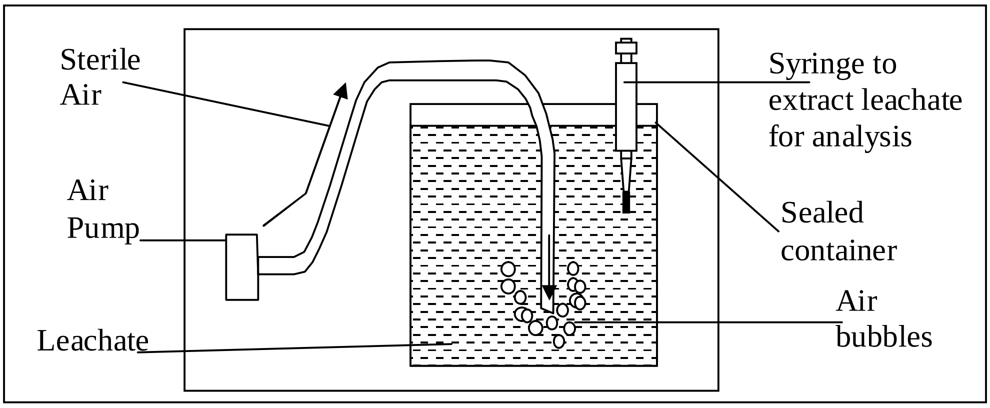


Figure 3.3: Set-up of the microbial treatment for leachate

Table 3.5: Series of leachate treatment trials with microbial cocktail.

Innoculum	Trials			
Bacterial cocktail	NB1 (100ml/L)	NB2 (100ml/L)	NB3 (100ml/L)	control
Fungal cocktail	PDB1(100ml/L)	PDB2 (100ml/L)	PDB3 (100ml/L)	control

3.2.6 Methane Oxidation Studies

Soil materials were collected from Panchang Bendena landfill from an active cell, an 8 months old cell, and 1 year old cell at different depth i.e. 10 cm, 20cm and 30 cm deep. The procedures in the methane oxidation studies were adopted from Scheutz and Kjeldsen (2001). Approximately 20 g of soil samples were placed in airtight Wheaton bottles, covered with a rubber stopper and sealed with aluminium cap as shown in Plate 3.21.

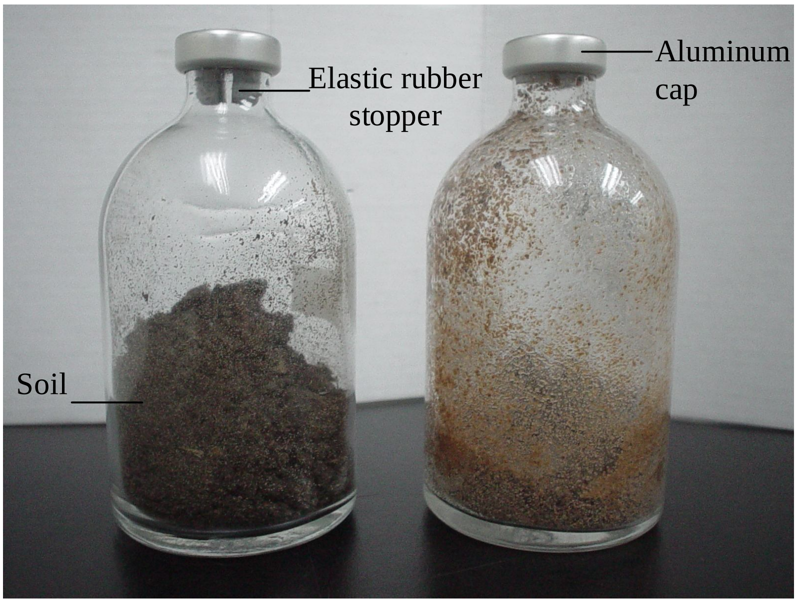


Plate 3.21: Set-up for methane oxidation studies

Thirty ml of air from the bottles were removed to be placed with 20 ml of oxygen and 10 ml of methane to determine the methane oxidation process in the soil samples incubated at 4°C to 35°C. Analyses were conducted using a micro gas chromatogram (GC) at various intervals to establish the optimum parameters for methane oxidation. The data were processed with Maestro Program.

3.3 Results and Discussions

3.3.1 Waste received at landfills

3.3.1.1 Waste Characterization/ Composition Studies

The highest percentage of MSW was organic waste. It ranged from 30% to approximately 60% of the total waste stream. It reflected a typical scenario observed among developing nations where organic waste contributed nearly half of the total waste stream (Zhu *et al*., 2009; Hao *et al*., 2008; Fauziah *et al*., 2004; Agamuthu *et al*., 2003; World Bank, 1999).

On average, paper and plastic waste contributed 13% and 15%, respectively which is also agreeable to the waste disposal trend in most developing countries (Zhu *et al*., 2009; Hao *et al*., 2008; Damanhuri and Padmi, 2000; World Bank, 1999). The percentage and density of the waste received at the nine landfills in Selangor are summarized in Table 3.6 and Table 3.7.

The bulk density of the waste is based on the data obtained from the study conducted at the nine landfills in Selangor where wastes were segregated into nine waste types. Waste density was measured based on the wet weight. Organic waste which made up the largest portion in the waste stream received by the nine landfills has an average density of 0.75 kg/m^3. The densities of other waste types were relatively lower than that of the organic waste, ranging from 0.19 to 0.32 kg/m^3. The higher the density of the waste, a more efficient landfilling process can be achieved. The typically low density of waste causes landfills space to be exhausted faster than the anticipated period (Narayana, 2009; Lilja and Liukkonen, 2008; Sha'Ato *et al*., 2007).

Table 3.6: Percentages (% on wet weight) of different waste types received by landfills in Selangor.

Waste types	**Panchang Bedena**	**Kpg Hang Tuah**	**Kundang landfill**	**Sungai Sedu**	**Ampar Tenang**	**Bukit Beruntung**	**Kerling**	**Hulu Yam Bharu**	**Sungai Kembong**
Organic	58.7 ± 2.3	51.6 ± 3.0	42.0 ± 2.1	50.0 ± 5.3	48.4 ± 0.9	45.8 ± 1.1	30.2±17.8	35.9 ± 0.8	58.0 ± 3.3
Paper	10.6 ± 4.5	10.9 ± 0.8	12.9 ± 2.2	16.1 ± 4.3	23.3 ± 5.6	18.7 ± 3.1	8.2 ± 1.7	17.4 ± 2.8	10.2 ± 4.1
Plastic	15.9 ± 4.1	18.6 ± 7.0	24.7 ± 5.1	13.8 ± 3.1	17.0 ± 2.2	12.7 ± 1.0	2.0 ± 1.8	15.5 ± 3.4	14.8 ± 2.0
Rubber	1.2 ± 0.8	3.0 ± 1.1	2.5 ± 0.3	6.4 ± 2.1	0.8 ± 0.1	2.6 ± 0.2	2.0 ± 0.8	7.1 ± 1.9	0.9 ± 0.1
Textile	3.6 ± 0.9	2.0 ± 0.3	2.5 ± 1.7	2.0 ± 0.4	1.3 ± 0.5	2.3 ± 0.5	3.1 ± 1.9	6.4 ± 2.2	1.4 ± 0.3
Metal	3.2 ± 2.1	3.9 ± 0.9	5.3 ± 2.1	2.7 ± 0.8	2.3 ± 0.3	3.6 ± 0.4	2.1 ± 0.1	2.9 ± 0.1	2.8 ± 0.9
Glass	2.2 ± 0.6	2.9 ± 1.2	1.8 ± 0.4	1.5 ± 0.6	2.4 ± 0.8	2.8 ± 0.7	3.9 ± 0.7	6.7 ± 0.3	2.7 ± 0.4
Wood	0.5 ± 0.1	2.4 ± 0.6	5.7 ± 1.7	1.2 ± 1.0	2.9 ± 0.1	2.3 ± 1.1	36.9± 12.5	5.9 ± 1.4	2.1 ± 1.0
Miscellaneous	4.2 ± 1.9	4.6 ± 0.1	2.5 ± 1.4	6.30 ± 2.3	1.7 ± 0.4	9.3 ± 0.3	11.6 ± 4.0	2.2 ± 0.8	7.2 ± 1.5

Table 3.7: Density (X 10^{-3} g/cm^3) of waste received by the landfills in Selangor

Waste types	**Panchang Bedena**	**Kpg Hang Tuah**	**Kundang landfill**	**Sungai Sedu**	**Ampar Tenang**	**Bukit Beruntung**	**Kerling**	**Hulu Yam Bharu**	**Sungai Kembong**
Organic	0.44 ± 0.1	3.94 ± 1.7	0.40± 0.01	0.20± 0.11	0.36± 0.01	0.31± 0.11	0.34± 0.09	0.34± 0.09	0.43± 0.23
Paper	0.19± 0.01	0.76 ± 0.6	0.08± 0.01	0.09± 0.02	0.12± 0.08	0.17± 0.04	0.21± 0.06	0.22± 0.11	0.08± 0.01
Plastic	0.15± 0.02	2.11 ± 0.9	0.21± 0.14	0.06± 0.01	0.01± 0.03	0.10± 0.02	0.07± 0.04	0.13± 0.03	0.07± 0.01
Rubber	0.2 ± 0.09	1.03 ± 0.1	0.10± 0.08	0.09± 0.06	0.14± 0.04	0.10 ± 0.01	0.16± 0.07	0.28± 0.23	0.35± 0.07
Textile	0.31± 0.19	0.64± 0.01	0.06± 0.02	0.11± 0.03	0.18± 0.07	0.09± 0.04	0.31± 0.14	0.41± 0.01	0.13± 0.04
Metal	0.22± 0.05	0.45± 0.03	0.05± 0.01	0.07± 0.01	0.06± 0.01	0.08± 0.01	0.12± 0.06	0.19± 0.04	0.07± 0.02
Glass	0.4 ± 0.1	0.2 ± 0.14	0.20± 0.13	0.05± 0.01	0.54± 0.01	0.21± 0.06	0.25± 0.08	0.04± 0.02	0.22± 0.16
Wood	0.33± 0.11	0.44± 0.12	0.04± 0.01	0.07± 0.01	0.06± 0.01	0.12 ± 0.10	0.48± 0.01	0.08± 0.03	0.12± 0.01
Miscellaneous	0.33 ± 0.1	3.43 ± 1.2	0.35± 0.07	0.10± 0.05	0.01±0.001	0.18± 0.04	0.32± 0.01	0.05± 0.01	0.17± 0.05

The main activities in the areas determine the quality and quantity of waste received by these landfills. It proves that types and quantity of waste disposed is closely related to its economic activities (Zhu *et al*., 2009; Fauziah and Agamuthu, 2006; Rathi, 2005; Agamuthu *et al*., 2004; Hoornweg, 2000; World Bank, 1999). Waste tonnage increased with the increase in population size and urbanization as indicated in Table 3.8.

Table 3.8: Types of disposal sites in Selangor

Landfill types	Locations	Average of waste received daily (tonnes)
Urban	Kundang landfill	400
	Sungai Kembong landfill	600
Sub-urban	Kampung Hang Tuah Landfill	100
	Sungai Sedu landfill	200
	Kerling landfill	150
	Ampar Tenang landfill	100
Rural	Panchang Bedena landfill	60
	Bukit Beruntung landfill	80
	Hulu Yam Bharu landfill	50

Urbanization plays an important role in the increase of waste (Minghua *et al*., 2009; Sokka *et al*., 2007; Henry *et al*., 2006) where urban landfills namely Kundang and Sungai Kembong landfills received higher tonnage of waste compared to the sub-urban and rural landfills. The highest waste contributors for rural and sub-urban landfills are the commercials centers and institutions. This is due to the shifting of industrialized and commercial zones from the urban to the sub-urban and rural sites to encourage economic development in the less urbanized area. On the other hand, largest waste contributors to urban landfills were the households. The rapid development to cater the need of urban population resulted with higher generation of domestic waste (Odum and Odum, 2006;

Haberl, 2006; Agamuthu *et al*., 2004). Table 3.9 depicts the percentage of waste from different generators namely domestic, commercials and institutions, and industries.

Table 3.9: Percentage of waste (on wet weight) from different sources

	Domestic	Commercial and Institutions	Industries	Total
Rural waste	38	40	22	100
Sub-urban waste	16	56	28	100
Urban waste	48	34	18	100

Industrial sector generated the lowest volume of waste in urban landfills. This could also result due to the increase in the price of land for residential and commercial activities that industries shift to sub-urban and rural areas. Waste from the commercial sector is high in the sub-urban and rural landfills. This is probably due to the larger number of commercial premises and institutions to accommodate the trading needs in these two areas. In addition to that, rural and sub-urban community in most areas disposed their garbage themselves by burning or burying. The domestic waste received in the sub-urban landfill was the lowest probably due to the urban migration factor which resulted with less population residing in the sub-urban area.

3.3.1.2 Detailed Waste Composition Studies

Food waste contributed the biggest percentage in the total waste composition which averaged at 52%. Detailed waste compositions from the three landfills are listed in Appendix 3.3-3.5. The percentages of food waste derived from each sectors namely industrial, commercial and institutions and residential were calculated to determine the average generation in three study areas. Table 3.10 summaries the percentages of food waste generation from the total waste stream by different sectors at the study areas.

Table 3.10: Food waste (% of wet weight) in the waste generated by different sectors by different areas.

	Average generation (areas) (%)	Food waste contribution by sectors (%)		
		Industrial	Commercial and Institutions	Residential
Urban	46	31	61	47
Sub-urban	17	4	19	28
Rural	25	1	40	35
Average (Institutions) (%)	29	12	40	37

On average, the largest contributor of food waste is the commercial and institutional sector at approximately 40% of the total waste generated in the study areas, followed by the residential sector at 37%. Similar findings were obtained in most developing countries (Zhu *et al*., 2009; Körner *et al*., 2008; Agamuthu *et al*., 2003; World Bank 1999). Figure 3.4 illustrates the trend in food waste generation from the three landfills by different income groups.

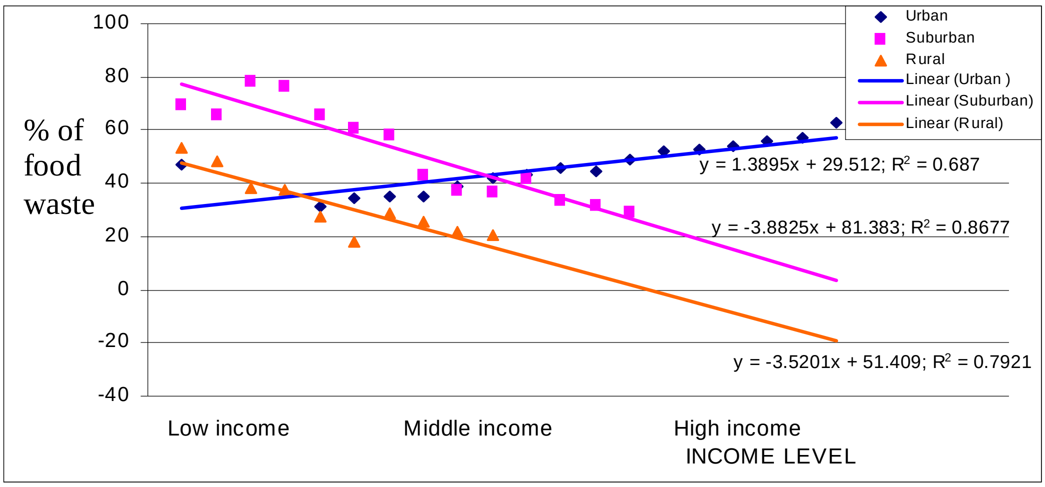


Figure 3.4: Food waste generation by different income levels at three landfills.

Food waste generation among the urbanites indicate an increasing trend (R^2=0.687) where high income groups generate more food waste than the lower income groups. The high income groups have a higher purchasing ability which allows them to purchase more food due to the abundance of food–selling premises in urban areas. As a result, the eating and throw-away habits among urbanites are more significant among the high income groups areas where more food waste are present in the high income groups than that of the low income groups. The findings are agreeable with the report that high income groups particularly in urban areas has tendency to waste more food than the lower income groups (Choy *et al.*, 2004; Irina and Chamuri, 2004). On the other hand, among the suburban and rural communities, food waste generation decrease with increase in income level with R^2=0.8677 and R^2=0.7921, respectively (Figure 3.4). This is closely related to the fact that there are fewer number of food-selling premises compared to that in urban areas. As a result, households have more tendencies to prepare food at home. The high income groups produces less waste (inedible portion) since they can afford processed and ready food items with minimum preparation from the consumers' end. The findings are agreeable with other reports from the food waste trend among suburbanites and rural consumers (Irina and Chamuri, 2004; Fauziah and Agamuthu, 2003).

The generation of not consumed food waste was found to increase significantly with the increase in income level in the rural (R^2=0.7682), sub-urban (R^2=0.8527) and urban (R^2=0.6572) landfills. It clearly indicates the existence of 'throw-away society' among Malaysians. This is agreeable with various findings by Choy *et al*. (2003) and Irina and

Shamuri (2004). Figure 3.5 depicts the increasing trends in not-consumed food waste among the different income levels.

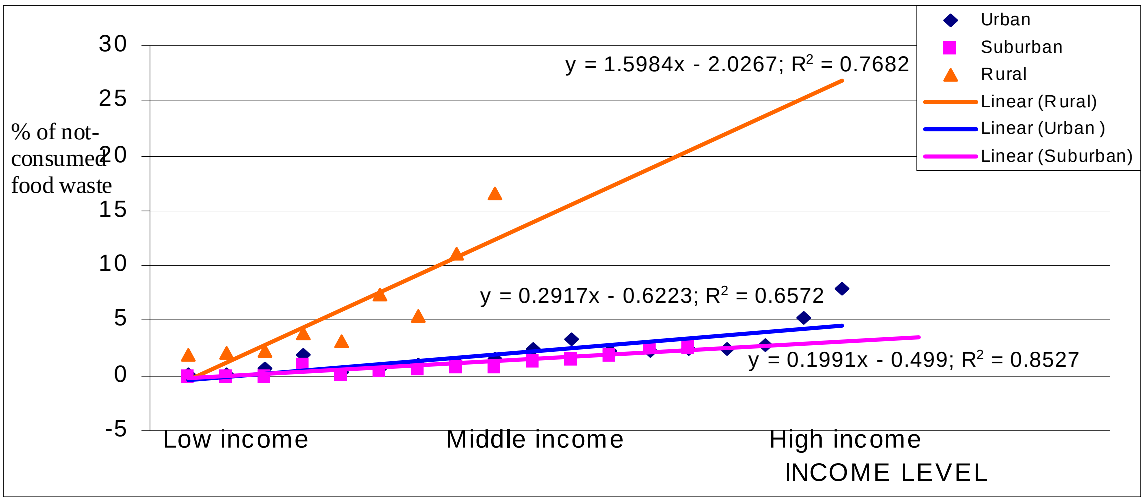


Figure 3.5: Increasing trend in not consumed food waste generation by the different income groups.

Total amount of paper waste received averaged at 14 -20% in the three landfills (refer to Appendix 3.5). The generation of newsprint averaged 10-14% in the three landfills. A decreasing trend was obtained among the urbanites where lower income groups were found to generate more of this waste than the higher income groups (R^2=0.9106). Figure 3.6 depicts the decreasing trend of newsprint generations by the different income groups.

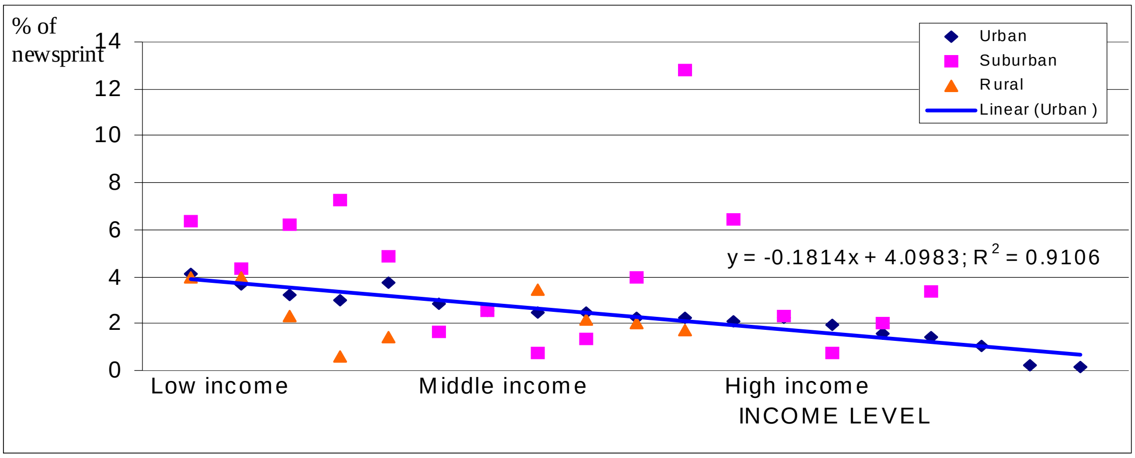


Figure 3.6: Decreasing trend in newsprint generation by different income group.

Daily newspaper subscription among higher income group result in consistent accumulation making recycling more economical and convenient. On the other hand, the lower income group which purchase newspaper at lower frequency accumulated the material at a too slow rate for recycling purpose. This resulted with constant disposal of newsprint into the low income waste stream. Approximately 74% of newsprint was received from the domestic source followed by 18% from the commercial sector in the suburban landfill. This is because newsprints were normally bought by individuals to their home to be read by the whole household. Though newspaper recycling is in practice, the unattractively low market price (RM0.20/kg) discourages public to recycle (Woodard *et al*., 2006; Perrin and Barton, 2001). The generation of magazine waste was low in all three landfills. This is probably due to the fact that readers who purchased magazine will not dispose this material but keep it as collection. In addition, take-back concepts by many printing companies resulted with low amount of this items being disposed off into landfills. A significant trend was observed among the rural groups (R^2=0.6302) where magazine waste increases with the increase in income level (refer to Appendix 3.6). It could be due to the higher affordability of the high income group to spend on magazine than that of the lower income group. On the other hand, recycling of these materials is low because of less market demand (Woodard *et al*., 2006; Alhumoud, 2005; Alhumoud *et al*., 2004). Similarly with corrugated paper at rural landfills which contributed 47% of the total paper waste stream. The commercial sector generates the largest percentage of corrugated paper due to the disposal of packaging wastes upon receiving bulky supplies. The trends indicate the increase in corrugated paper waste generation with the increase in income level in all landfills. Figure 3.7 illustrates trends observed for corrugated paper generation.

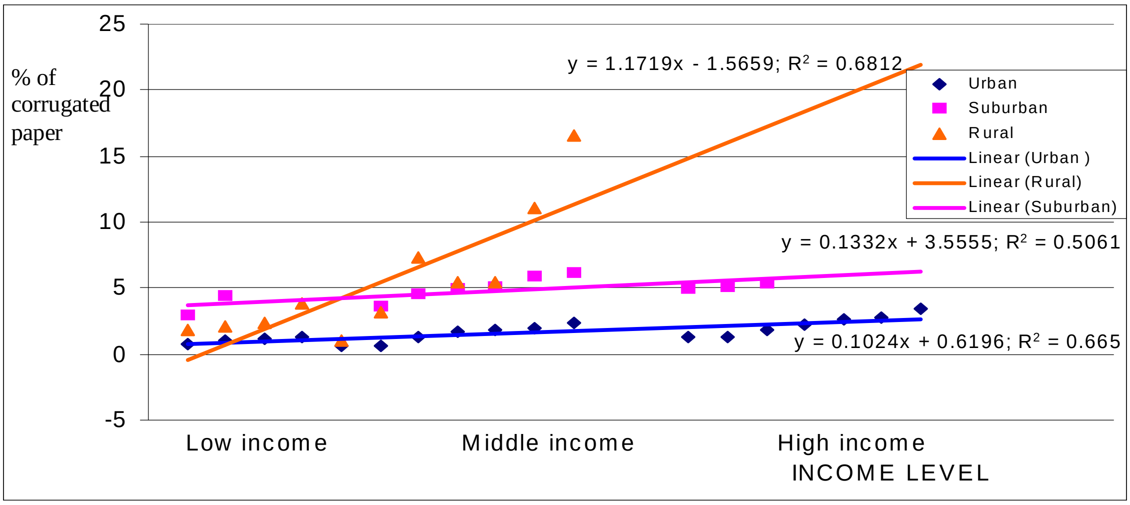


Figure 3.7: Increasing trends in corrugated paper (waste) generation by different income groups.

The ability to purchase goods increased with increase in income level resulting with more packaging material including corrugated papers. The lack of 'take-back' practice by many retailers and suppliers disrupt the recycling options of many packaging material including corrugated papers. Studies indicated increasing recycling rate can be achieved when more convenient 'take-back' services are provided to the consumers (Alhumoud, 2005; Alhumoud *et al*., 2004; Grodzińska-Jurczak, 2001). This will eventually reduce its amount from entering the waste stream. Mixed paper contributed approximately 34-74% of the total paper waste in the three landfills with rural landfill receiving the highest percentage (74%). A low amount of white paper indicates the possibility that the material was channeled out of the main waste stream. It was reported by George and Agamuthu (2003) that the recycling of this component is most likely the reason for its low disposal rate due to its slightly higher market price in addition to the fact that Malaysians' paper consumption was recorded at 75kg/per capita (Earthtrend, 2002).

The second biggest percentage of waste generated by the three landfills is plastic waste which contributed approximately 13-38% of the total waste stream (Appendix 3.3). The highest percentage of plastic component is films plastics (33-44%) mainly of plastic bags. It is due to its abundant use in the commercial sector particularly in the urban areas. Highest film plastic waste generators in suburban and rural landfill were the industrial sector contributing 50-60% of the total plastic waste received. Recycling of film plastics is not in practice due to the unattractive incentives and its' low market price (Najafi *et al*., 2006). Among the domestic waste generators, plastics were abundantly used because it can be obtained easily and cheap (Najafi *et al*., 2006; Zabaniotou and Kassidi, 2003; Subramanian, 2000; Rao, 2000). In addition, plastic bags are commonly reused to keep waste for disposal purpose. The disposal of film plastics was significantly correlated with the income level of the urban (R^2= 0.6363) and sub-urban groups (R^2= 0.8216) as shown in Figure 3.8.

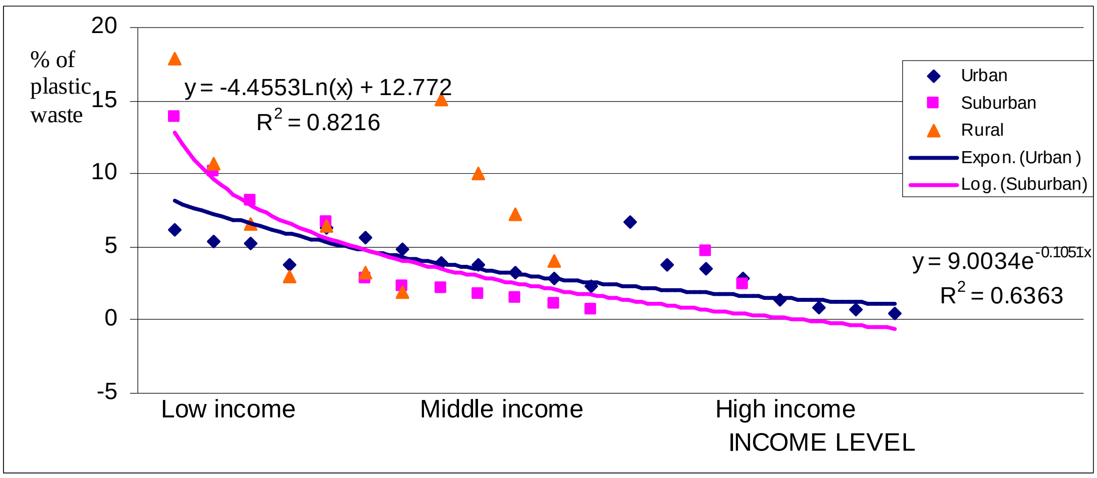


Figure 3.8: Film plastic waste generation from different income level.

Figure 3.8 shows decreasing trends in film plastic waste generation with the increase in income level. Since plastics are very convenient (Agamuthu *et al*., 2006) it is widely available in the market and it is used to bag goods by traders and retailers. The more available plastics bags are to the consumers, the higher is its disposal into the waste stream (Fauziah and Agamuthu, 2003). On the other hand, goods of higher standard which can be afforded by the higher income groups normally come with stylish paper bags. This resulted with high income groups having to dispose less plastic bags into the waste stream (Zabaniotou and Kassidi, 2003). Unlike film plastics waste, polystyrene wastes increased in all three landfills. It increased significantly with the increasing income level in urban (R^2=0.5755), suburban (R^2=0.7814) and rural (R^2=0.6317) landfills, as shown in Figure 3.9.

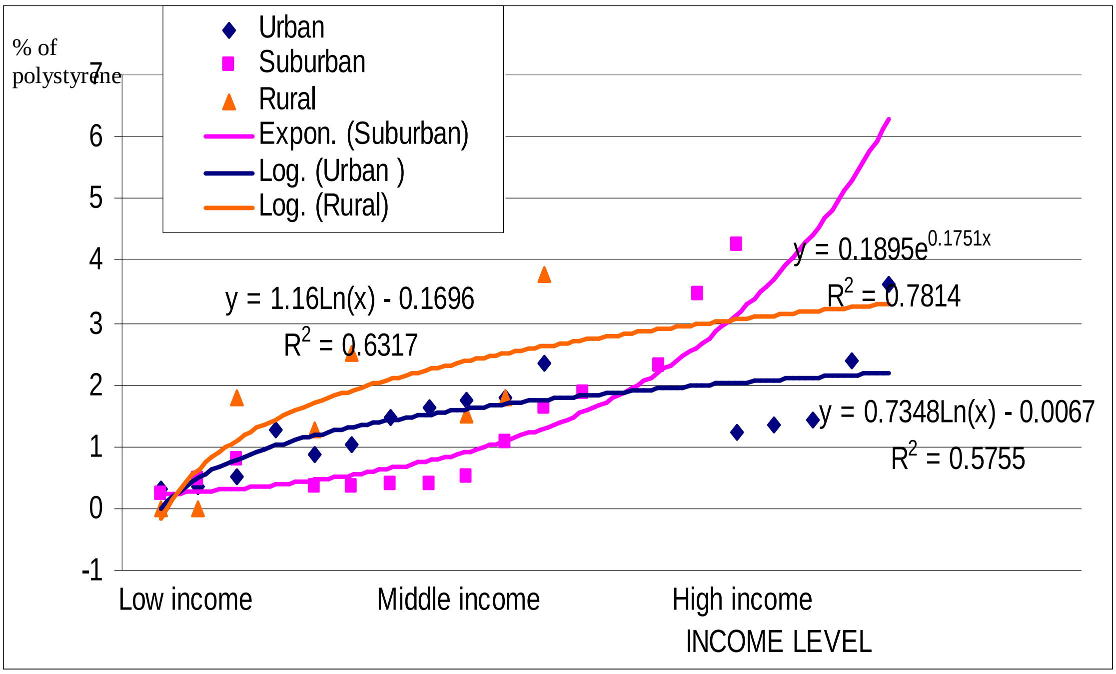


Figure 3.9: Increasing trend in polystyrene waste generation by different income groups.

Studies indicated that consumer with higher living standard demand higher quality and more convenient goods with less preparation on the consumers' end regardless of its high price (Odum and Odum, 2006). Current packaging practice was aimed to ensure consumer satisfaction that polystyrene is widely utilized. This could also be attributed by the high utilization of polystyrene food packages (Demirbas, 2004; Zabaniotou and Kassidi, 2003) among the higher income group that preferred the take-away option. Disposable diapers comprised of approximately 26%, 6% and 9% of the total plastic waste generation in urban, sub-urban and rural landfill, respectively. Similarly, the generation of disposable diapers showed increasing trend (Appendix 3.7) in urban (R^2=0.7047) and sub-urban (R^2=0.5672) landfills. Increase in income level resulted with increase in disposal of disposable diapers. Convenience is the main factor resulting with more utilization of disposable diapers in high income household besides the affordability factor. In sub-urban and rural landfill, disposable diapers were mainly generated by the commercial sector (57-64%) followed by households. This is because disposable diapers' users spend less time at home that the generation was reduced while institutions such as nurseries, childcare centers and old folk homes contribute bigger amount. In addition to that, many rural households still prefer washable cotton diapers than the disposable type which reduces the percentage of disposable diapers in the waste stream.

Rigid plastic was the highest and second highest in sub-urban and rural landfills, respectively, contributing 20-46% of the total plastic waste stream. Industries were the main source of its generation in the urban and rural landfill due to the high utilization of this material in the manufacturing process. This is because plastics are easily obtained and cheap (Demirbas, 2004; Subramanian, 2000; Rao, 2000). Trends in the generation of rigid plastic waste are indicated in Figure 3.10.

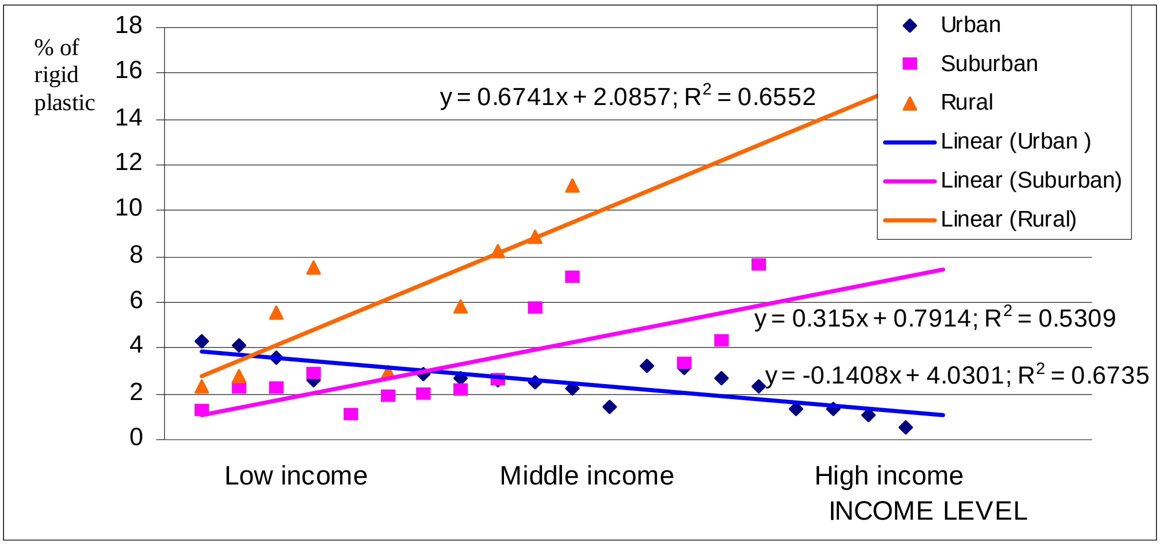


Figure 3.10: Rigid plastic waste generation by different income groups

Increasing trends for rigid plastic were observed in sub-urban and rural landfills with R^2=0.5309 and R^2=0.6552, respectively. This is due to the increased utilization of rigid plastics as income level increased. The abundance of rigid plastic in the market resulted with more of its utilization by the consumers particularly of the higher income groups in rural and sub-urban areas since it is more convenient and affordable. On the other hand, among the urbanites, generation of rigid plastic waste decreased with increase in income level. This is probably due to the awareness on the detrimental effects of plastic to the environment through various campaigns in the urban areas. In addition to that, an increase in the standard of living resulted with consumers demanding more environmental friendly goods (Odum and Odum, 2006) besides plastics.

Textile waste received by the landfills was mainly generated by the industrial (67%) and commercial sectors (22-77%). A significant (R^2=0.5469) increasing trend was observed between textile waste and income level among sub-urban households (Appendix 3.8) It is contributed by increase in affordability as income level increased to dispose these unwanted goods. The "throw away" attitudes are more evident among the high income people than those of the lower income (Fauziah and Agamuthu, 2007; Demirbas, 2004; Grodzińska-Jurczak, 2001).

Approximately 5-8% of the waste stream consisted of garden waste where 42% was generated by the residential areas. The lack of space to allow natural degradation and composting resulted with its disposal into the MSW stream. If these components can be diverted for composting, it can produce good quality compost as achieved in North of Belgium (Leroy *et al*., 2007). Appendix 3.9.depicts the garden waste increasing generation trends with the increase in income in sub-urban (R^2=0.5564) and rural (R^2=0.6129) landfills. The increase in household space and larger house compound among the higher income groups resulted with more plants and vegetations that require trimming. On the other hand, the lower income groups possess less space for gardening purpose. These findings are agreeable with those of Choy *et al*. (2004) and Irina and Chamuri (2004) where high income groups generated more garden waste than the lower income groups.

The generation of aluminium cans were significantly (R^2=0.616) related to the income level of the urban waste generators where higher income groups generates more of this material than the lower income groups (Appendix 3.10). The trend is contributed by the fact that goods particularly food are more expensive when it is packed in aluminium cans which is less affordable by the lower income group. In addition to that, the high market price of aluminium cans has probably diverted this component for recycling purpose which led to its scarcity in the lower income groups' waste stream. Similar findings were found that more of the lower income people tend to recycle components which have high market price such as aluminium cans and other metals (Batool *et al.*, 2008; Pappu *et al.*, 2007; Damanhuri and Padmi, 2000; Leu and Lin, 1998).

Hazardous waste was approximately 0.5% of the total waste which was mainly sourced from the household waste stream (56%). Correlation between hazardous waste generation and income level was obtained in all three landfills as illustrated in Figure 3.11.

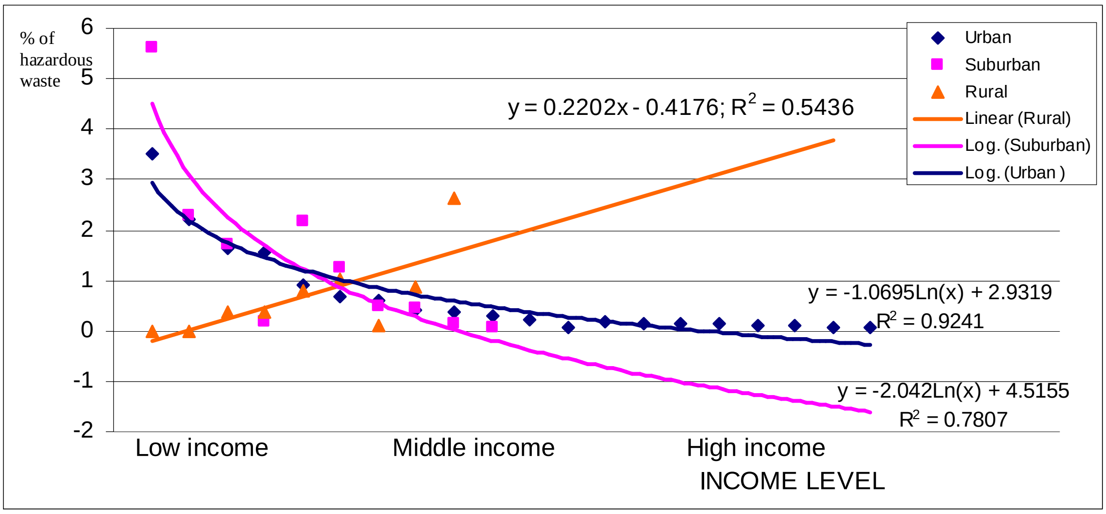


Figure 3.11: Significant trend in hazardous waste generation by different income groups.

In urban (R^2=0.9241) and suburban (R^2=0.7807) landfills, hazardous waste generation indicated logarithm regression. Higher income groups generate lesser amount of hazardous waste as compared to the lower income groups. This could be attributed to the increased awareness, due to the intensive campaigns launched in the cities that less hazardous waste were disposed off into the waste stream. In addition, the establishment of collection centres by Non-governmental Organizations (NGOs) in cities enables people to send their hazardous waste for proper disposal. The reduction could also due to the affordability among the higher income consumers to purchase environmental friendly product (Odum and Odum, 2006). However, the higher income groups in rural areas disposed more hazardous waste compared to the lower income groups. No other alternative for household hazardous waste disposal exists for the rural community. In addition, the lack of policy concerning hazardous waste disposal from household also resulted with its presence in the waste stream, as observed in previous studies (Duan *et al*., 2008; Bhattacharyya *et al*., 2006). Only 8% of the hazardous waste received in the MSW stream were sourced from industrial sector. This is due to the industrial compliance to regulations pertaining to industrial hazardous waste disposal. The 8% generation probably sourced from irresponsible manufacturers of the small and medium industries. The absence of monitoring and enforcement by the appropriate authorities allow these sectors to find easier and cheaper way to be rid of their hazardous waste. Various reports detailed the problem faced by many municipalities in Malaysia to curb indiscriminate hazardous waste dumping at secluded area and into the MSW stream.

The generation of clear (R^2=0.670) and coloured glass (R^2=0.6113), and bulky waste (R^2=0.6537) in the sub-urban increased with the increase income level (Appendix 3.11 - 3.13) while no significant trends were observed among the urban and rural groups. The positive increase was probably due to the affordability of higher income groups in sub-urban areas to purchase processed goods particularly food items in bulk, which are packed in glass containers to maintain its freshness. Various studies indicated the tendency of consumers to purchase higher quality products increased with the increase in living standard and affordability (Odum and Odum, 2006; World Bank, 1999). Rubber waste contributed approximately 1.2% of the total waste stream. Domestic source accounted 27-41% of the total rubber waste received in all three landfills. Appendix 3.14 illustrates the increasing trends observed in rubber waste generation with increase income level at urban (R^2=0.5648), sub-urban (R^2=0.6506) and rural (R^2=0.6181) landfills. This could be attributed by the 'throw-away' habits particularly among the higher income consumers. Their affordability to purchase new goods allowed them to indiscriminately disposed unwanted items more often than that of their lower income counterparts (Choy *et al.* 2003; Irina and Chamuri, 2004). The consecutive paragraphs discuss waste generation data on randomly selected households of different income level in urban, suburban and rural area with the inclusion of fasting month and non-fasting months sampling.

As a conclusion, organic waste made up the largest portion of the waste received by the landfills. Non-consumed food, polystyrene and corrugated paper were discarded more by the higher income groups where increasing trends were obtained while the generation of other waste types differed between the different income groups.

3.3.2 Characteristics of MSW Generated (At Source)

The complexity of waste generated was found to vary with the different levels of urbanization. The average quantity of waste generated by the three different areas is shown in Appendix 3.15. Analyses on the results of the waste generation studies are discussed in the consecutive sections.

3.3.2.1 Urban Households

Urban areas are known to generate higher amount of waste volume due to the improved living standard, as well as, the high per capita income (Odum and Odum, 2006, Agamuthu *et al.*, 2004; Agamuthu, 2001). Appendix 3.16 details the results of waste (fresh weight) generated by different income groups in urban areas while the per capita generation of MSW in the urban area during fasting month and normal period is depicted in Table 3.11.

Table 3.11: Average waste generation (kg/capita/day) by different income group in urban area.

Household income	Non fasting months	Fasting month	Increase in waste generation (%)
High income	1.74 ± 0.43	1.97 ± 0.75	13 %
Middle income	1.11 ± 0.16	1.32 ± 0.13	19 %
Low income	1.12 ± 0.12	1.24 ± 0.44	11 %
F-test	0.334271		

Results indicate that the highest quantity of waste was generated by the high income group at approximately 1.7 kg per capita, followed by both the low and middle income groups at 1.1 kg per capita. The affordability to consume more could result in higher waste generation. It has been reported that economic status influenced MSW generation capacity in any area (Troschinetz and Mihelcic, 2009; Shekdar, 2008). A slight increase was observed during fasting month which could be contributed by the consumption habit of the Muslim community (60% of the nation's population) and a small portion of the non-Muslim who tend to spend more during fasting month. This has been reported in various socio-economic studies that Malaysian tend to spend more money during the month of Ramadhan (Dusuki, 2008). The average MSW per capita generation during fasting month was highest among the high income groups at approximately 2.0 kg MSW followed by the middle and low income groups at 1.3 kg per capita and 1.2 kg per capita, respectively. However from the F-test conducted, there is no statistical significance of these differences in waste generation between fasting and the non-fasting months.

On average, the total organic portion of the waste excluding paper was 41% with food waste ranging from 21% to 26% (Appendix 3.16). Approximately 4% was not consumed food which was largely contributed by the high income group (4.2%). The higher capability to purchase goods particularly food, tend to encourage food wastage. This is agreeable with an online community poll of 3,942 votes which indicates that respondent admitted to purchase more than necessary as stock and 31% agreed that they disposed not consumed food when it reaches expiry (Care2, 2008).

Approximately 20% of the total MSW generated by the urbanites consist of paper wastes where the low income groups contributed the largest (36%) followed by the middle income (35%) and high income (29%) groups. The unattractive market price for paper recycling could be the reason why it disposed, contributing as approximately 1/3 of the low and middle income groups' waste stream (George and Agamuthu, 2003). On the contrary, plastic generation was highest from the middle income groups (38%) which exceed 28% and 13% of that generated by the low and high income groups, respectively. Abundance of plastic products resulted with its high disposal since it is averagely affordable (Manga *et al*., 2008; Choy *et al*., 2003).

Generation of disposable diapers indicate an increasing trend as the income level decreases where the low income groups contributed the highest percentage (50%) with approximately 49% higher than the average generation in the urban area. This could be attributed by the fact that the low income groups normally provide cheap child care service that is affordable for the middle income working parents.

Approximately 2.5% of the urbanites waste consisted of textile waste while 8% and 1% comprised of garden and rubber wastes, respectively. Lower income groups generate the lowest percentage (6%) of garden waste and the percentages increase with the increase in the income level. The high income groups generated 39% more garden waste than the average urbanites generation. This generally could be attributed by the larger gardening space among the high income groups compared to the middle and low income groups which have limited or no space for gardening.

On average, 8.3% of the urbanites MSW consisted of coloured and clear glasses, with the low income group being the largest contributor (38%). The presence of coloured glass was higher in the MSW stream at an average of 4.6% compared to clear glass (3.7%).

The low income groups contributed the highest portion of clear glass (39%) i.e. 20% more than the average generation among the urbanites. This is probably due to the habit of purchasing more goods particularly food products in coloured glass containers than in clear glasses. In addition to that, the low market price for coloured glass recycling probably discourages the diversion of this component from the low income MSW stream.

Metal made up 15% of the urban MSW generation. The largest contributor (38%) was the middle income groups which exceeded 14% of the average generation followed by the high income and low income group. Tin-based waste was the largest components (38%) found within the metal categories followed by aluminum cans (23%) and other metals (21%). The middle income groups generated the highest percentage (41%) of aluminum cans followed by the high (32%) and low (28%) income groups. This is due to the lack of recycling of aluminum cans among the middle income group because of inconveniences. This is agreeable with the previous findings that inconvenience in terms of insufficient recycling center hinders the diversion of recyclable materials from the waste stream (Martin *et al*., 2006; Fauziah and Agamuthu, 2006).

Hazardous waste contributed an average of 2.5% of the urbanites MSW with high income group as the largest contributor (38%) followed by the middle and low income group at 33% and 29%, respectively. Similar observations were reported in many developing countries in regards to the presence of hazardous waste component in the MSW stream (Musson and Townsend, 2009; Fauziah and Agamuthu, 2008; Delgado *et al*., 2007). The presence of these components in the MSW stream resulted from the absence of proper disposal system for hazardous waste generated by the households.

To conclude, types of waste mainly generated by the urbanites are organic waste, paper and plastics. The per capita waste generation in urban area was found to exceed the average waste generation in the country.

3.3.2.2 Sub-urban Households

The high income groups generated the biggest percentage of waste as compared to middle and low income groups. At 1.72 kg per capita, the high income groups exceeded 51% and 105% of the average per capita generation of waste by the middle and low income, respectively. Similar to other area, the generation of waste differs between the fasting and non-fasting months. Table 3.12 shows the average waste generation in the sub-urban during the fasting and non-fasting months.

Table 3.12: Average waste generation (kg/capita/day) by different income group in the sub-urban.

Household income	Non fasting months	Fasting month	Increase in waste generation (%)
High income	1.72 ± 0.32	2.12 ± 0.71	23 %
Middle income	1.14 ± 0.12	1.23 ± 0.31	7.9 %
Low income	0.84 ± 0.27	0.91 ± 0.22	8.3 %
F-test	0.256751		

During fasting months, waste generation increased on average of 13%. The largest increase was among the high income groups which exceed 77% of the averaged increase among the sub-urban dwellers. The increase is mainly due to the changing consumption habit during the fasting month. However, results from the F- test indicated that there is no significant difference in the waste generation during fasting and non-fasting months.

The generation of different waste types by the sub-urban residents was slightly different from that of the urbanites. Sub-urban residents generated very high food waste which contributed an average of 59% with low income and the middle income groups both contributing 61% (Appendix 3.17). This could be attributed by the utilization of more not-processed food materials that the wastage during the process of preparing food was higher. Not-consumed food waste generation averaged at 0.9% with the high income being the largest contributor (50%) followed by the low income group (35%). This probably is due to the typical habit of stacking monthly stocks at home and buying at large quantity among the dwellers in the sub-urban corresponds with the findings among the urbanites. Appendix 3.17 depicts the average percentages of waste generated by the high, middle and low income group in the sub-urban areas.

The generation of paper waste ranged from 9% to 11%, largely consisted of corrugated paper. The middle income group generated the largest percentage (37%) of corrugated paper followed by the high income groups (33%). On the other hand largest contributor of newsprint was the high income groups at 39% followed by the middle income at 37%. Plastic waste generated by the sub-urban resident averaged at 6% with disposable diaper making up the largest percentage (34%) followed by rigid (23%) and film (23%) plastics waste. This generally could result from the abundant usage of plastic materials among household dwellers due to its cheap price (Manga *et al.*, 2008). The high income groups contributed 39% of plastic waste followed by middle (35%) and low (26%) income. Rigid plastics totaling to 35% were the main plastic waste generated by the high income followed by film plastic (29%). On the other hand, the largest amount of plastic waste generated by the low income group was plastic film (42%). The abundant use of plastic bags in daily life has encouraged its disposal into the waste stream.

Approximately 59% of garden waste generated in the sub-urban area originated from the high income groups, a similar trend observed in the urban area. This is largely due to the types of dwelling of the high income groups which offer more area for gardening and greeneries as compared to the middle and the low income groups. As for glass waste, the highest generators were the middle income group which contributed 41% of the total glass waste generation in the sub-urban areas. Generation of metal-based waste ranged from 4% to 6% with high income and low income groups being the biggest generators. The low generation of metal by the middle income groups could be explained by their recycling participation as compared to their low and high income counterparts due to environmental awareness and economic factors, respectively (Refsgaard and Magnussen, 2009; Troschibetz and Mihelcic, 2009). Hazardous wastes made up 1% of the MSW generated by the sub-urban residents. It included aerosol cans, insecticides, detergents, batteries and pharmaceutical products due to the absent of a proper disposal system in the country.

The main types of waste generated by the sub-urbanites are food waste, paper and plastics. On average, the per capita MSW generation by the sub-urbanites was below the Selangor average.

3.3.2.3 Rural areas

In the rural area, the average waste generation was 1.2 kg per capita with the high income group exceeded 23%. Generation by the low income groups were 31% lower than the average waste generation. Table 3.13 indicates the average waste generation by the rural household.

Table 3.13: Average waste generation (kg/capita/day) by different income group in the rural.

Household income level	Non fasting months	Fasting month	Increase in waste generation (%)
High income	1.51 ± 0.22	1.56 ± 0.41	3.3 %
Middle income	1.35 ± 0.61	1.21 ± 0.47	-11.6 %
Low income	0.85 ± 0.16	0.83 ± 0.23	-2.4 %
F-test	0.946659		

Increase in waste generation during fasting month is only evident among the high income groups while the middle and low income indicated decreases. The low affordability probably is the main factor that resulted with reduction in waste generation during the fasting month. However statistically (F-Test), no significant difference was derived.

In the rural area, food waste contributed an average of 57% of the total waste generated with the middle income groups being the biggest generator (35%) followed by the low and high income groups at 33% and 32%, respectively. High income generates the lowest percentage of food waste probably due to the utilization of processed food that food wastage was minimal. Appendix 3.18 details the average percentage of waste generated by the rural households.

The generation of paper waste ranged from 10% among the high income to 13% among the middle income groups. The low income groups generated 11% of paper waste with the highest portion (45%) contained mixed paper which is also the biggest contributor of mixed waste among the rural dwellers. The middle income groups were the largest contributors of corrugated paper and newsprint covering 47% and 39% of the materials, respectively.

The largest plastic waste generated was disposable diaper (42%) followed by plastic film (27%). Plastic film was mainly generated by the low income group totaling 41% of the total plastic film generated by the rural dwellers. Disposable diapers generation was highest among the high income groups (42%). Similar to the generation of garden waste in the urban and sub-urban, the biggest generator was the high income groups which exceed 48% of the average garden waste generation (2.7%).

The high income groups generated the largest amount (47%) of metal-based wastes followed by the low (33%) and middle (21%) income groups. Aluminum cans were generated mainly by the high income group which contributed approximately 65% of the total aluminum cans generated by the rural dwellers.

The main type of waste generated by the rural dwellers is food waste. Main composition of plastics waste generated in this area is disposable diapers. On average, the per capita MSW generation by the rural community was below the Selangor average.

3.3.3 Waste Analysis

In order to conduct waste analysis, at least five MSW samples were collected from each of the nine landfills in Selangor to obtain the average and the standard deviation. On average, moisture content of the waste samples was approximately 58%. It ranged from 38% in Sungai Kembong to 72% in Ampar Tenang. The results of waste analysis obtained from the respective landfills are summarized in the following Table 3.14.

Table 3.14: Chemical Analysis of MSW from Selangor Landfills

Landfills	**Moisture (%)**	**pH**	**Conductivity (μS/cm)**	**Salinity (g Na /kg)**
Sg. Besar Landfill	59.80 ± 20.37	5.95 ± 0.67	597.96 ± 37.05	0.78 ± 0.46
Kampung Hang Tuah	62.23 ± 1.82	5.35 ± 0.7	566.8 ± 238.58	0.68 ± 0.62
Sungai Sedu	64.16 ± 22.9	5.53 ± 0.64	1488.57 ± 27.04	1.04 ± 0.08
Ampar Tenang	72.27 ± 10.55	4.51 ± 0.27	857.49 ± 60.93	0.94 ± 0.04
Kundang	50.86 ± 15.73	5.42 ± 0.87	874.67 ± 133.24	1.40 ± 0.94
Kerling	66.58 ± 12.63	5.65 ± 0.77	856.67 ± 60.78	0.43 ± 0.03
Hulu Yam Bharu	56.42 ± 20.39	5.37 ± 0.98	1212 ± 61.2	1.02 ± 0.33
Bukit Beruntung	55.72 ± 24.89	6.62 ± 1.60	1104 ± 76.0	0.64 ± 0.05
Sungai Kembong	38.44 ± 1.0	5.42 ± 0.92	165.05 ± 83.13	0.9 ± 0.07

The high moisture content indicates a typical characteristic of waste generated by a developing country. It was found agreeable to the studies by Liu *et al.* (2008) and Sharholy *et al.* (2008) in China and India, respectively. The high moisture content could be attributed by many factors. Among others is the improper waste storage prior to collection and disposal to landfill which allow rain water to be trapped in the waste. Also, the typical habit of throwing everything including liquid waste into the waste bin among Malaysian caused the increase in the moisture level.

The wastes analyzed were acidic ranging from pH4.5 to pH6.6. The acidic characterization of the waste is probably contributed by the acidic components generated during the early stage of waste degradation or immediate rapid degradation by the microorganism within the waste. With sufficient moisture content and the appropriate temperature, the breakdown of the organic component in the waste will occur at a rapid stage resulting with the production of acidic compounds by the decomposers (Hao *et al*., 2008; Kruempelbeck and Ehrig, 1999). The lowest conductivity value (0.16 mS cm^{-1}) was obtained from the waste sampled from Sungai Kembong landfill while the highest value at 1.5 mS cm^{-1} was from Sungai Sedu landfill. The findings were very much lower than the conductivity value at 15 mS cm^{-1} obtained by Renou *et al*. (2008). The low hydraulic conductivity was found to reduce the flow of solution particularly leachate through the landfills layer (Cokca and Yilmaz, 2004). Salinity ranged from 0.43 gNa kg^{-1} to 1.4 gNa kg^{-1}. This was much lower than the value obtained in other studies (Fan *et al*., 2006; Hicklenton *et al*., 2001; Gil *et al*., 2000). This could be attributed by the high moisture content in the waste which dilutes the salinity.

3.3.4 Waste treatment Options

3.3.4.1 Vermi-composting

The first trial allowed the identification of possible problems which would occur if vermi-composting is practiced at the household level. Insects which breed in the vermi-composting system cause food competition (Plate 3.22). Table 3.15 lists the difficulties faced using vermi-composting set-up 1 (VCS1) and possible counter-measures.

Table 3.15: Problem arises from VCS1.

Factors	Problem	Counter-measure
Vermi-composting set-up	1. Worms escaping from pot 2. Insect breeding in the vermi-compost set-ups (Plate 3.22)	Cover the pot with suitable cover, e.g. porcelain flower pot-base
Surface/cover soil	1. Fungal growth (Plate 3.22)	Constant mixing of surface soil to prevent rapid growth of fungus
Moisture level	1. surface soil dry-up easily	Cover material used should have glossy surface to allow evaporation drip back into the system.

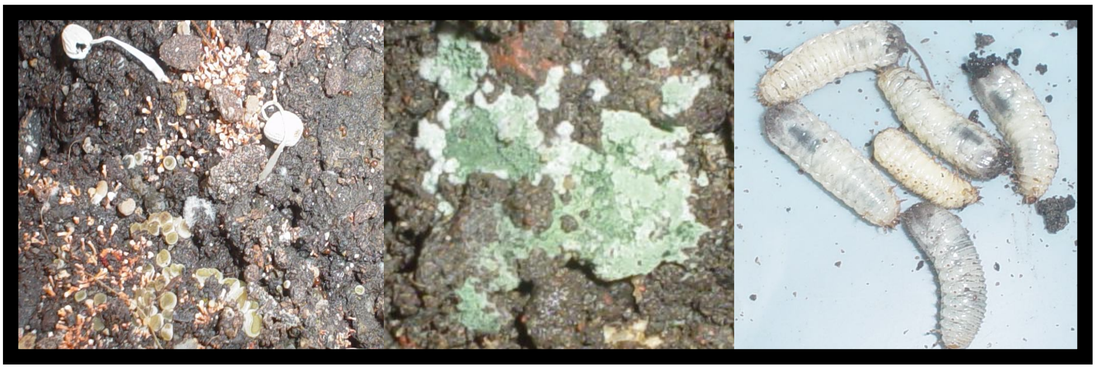


Plate 3.22: Fungal and insect problem in VCS1.

The rate for complete conversion to vermi-compost varied with the additives. The fastest was KW+VC (19 days), followed by KW alone (21 days). KW+GM and KW+GS were completed after 24 days while KW+GC took the longest period i.e. 60 days. This is slightly different to the rate obtained in the second vermi-compost set-up (VCS2). Percentage difference in time required for complete vermi-composting in VCS1 and

VCS2 is shown in Figure 3.12. The results of the analysis of vermi-compost in VCS1 is depicted in Table 3.16 and Table 3.17.

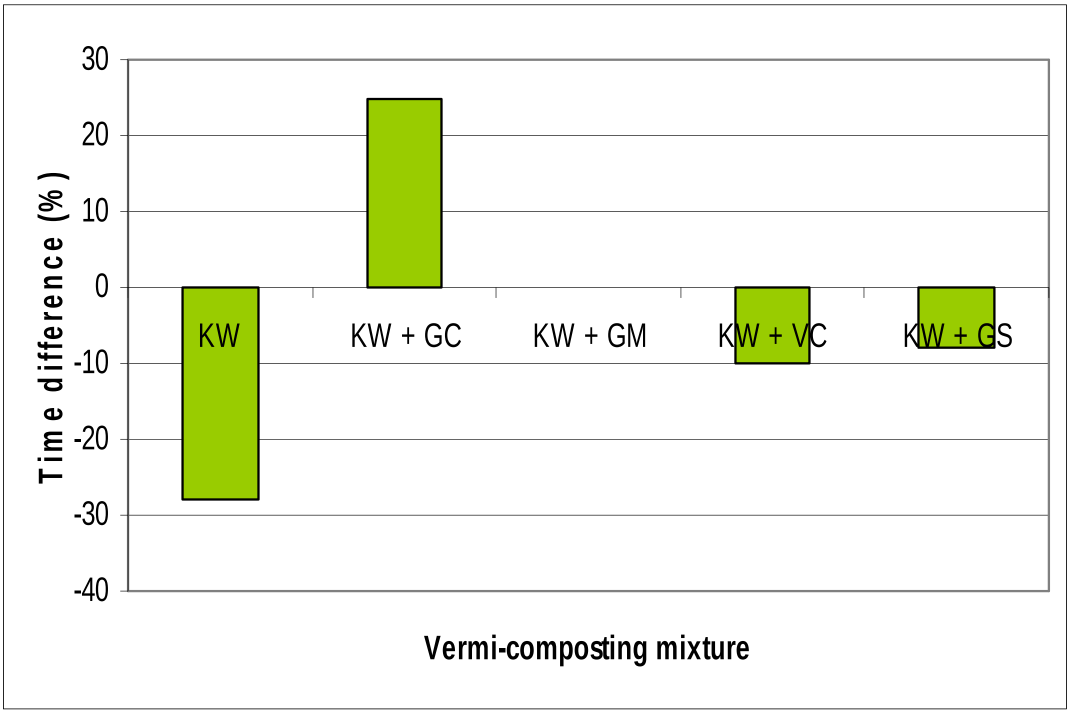


Figure 3.12: Differences (%) in completion time between VCS1 and VCS2.

Table 3.16: Analysis of vermi-compost in VCS1.

Combination	pH	Water holding capacity (%)	Total organic carbon (%)	Nitrate (mg/kg)	Phosphate (mg/kg)	Total Potassium (mg/kg)
KW	5.2±0.3	63.8±2.1	10.3±0.6	24.2±1.7	19.5±2.0	17.7± 1.5
KW + GC	6.1±0.1	59.2±0.9	11.2±0.3	18.5±0.2	44.6±0.4	9.0± 0.2
KW + GM	6.9±0.9	68.3±1.3	12.1±0.2	23.4±1.8	223.4±11.9	26.3±1.4
KW + VC	6.5±0.7	71.4±1.1	11.8±0.7	22.2±0.7	34.8±2.7	14.6±1.2
KW + GS	6.8±0.4	69.4±0.8	10.0±0.7	5.5±0.1	56.1±3.1	7.3±0.4

Table 3.17: Results of metal elements in vermi-compost (mg/kg) in VCS1.

Samples	Cr	Cu	Cd	Hg	Ni	Zn	Pb
KW	nd	0.01±0.001	nd	9.2±1.7	11.3±0.4	6.1±0.4	nd
KW+ GC	nd	0.01±0.004	nd	8.7±2.2	10.3±0.2	15.3±2.4	nd
KW+ GM	nd	0.01±0.006	nd	9.1±0.4	9.4±1.7	14.5±1.1	nd
KW+ VC	nd	0.03±0.001	nd	4.1±0.4	7.3±1.3	44.1±2.9	nd
KW+ GS	nd	0.02±0.005	nd	8.7±1.1	11.2±0.8	20.1±2.4	nd

Note: nd = not detected

Time required for VCS2 was higher for KW, KW+VC and KW+GS. This was generally due to the absence of unwanted organisms that organic matters are available for a longer period of time for the worms to use. The system with covers has less problems of insects and fungal growth. The vermi-composting process was 8-28% longer since the closed system prevented unwanted organisms from disturbing and competing for food with the worms. In both set-ups, KW+VC was the fastest to be completely converted into vermi-compost (21 days). The addition of vermi-cast created a more suitable environment for the worms to allow faster vermi-composting process due to the abundance of bacteria within the system (Padmavathiamma *et al*., 2008). It also promotes the development of juvenile worms (Plate 3.23) which enhanced the vermi-composting process.

Plate 3.23: Juvenile worms collected in the vermi-composting plates.

Completion of vermi-composting is determined with the absence of the initial organic waste i.e. KW, GM, and GC. The vermi-composting for KW+GM and KW+GS completed after 24 days and 26 days, respectively. Similar conclusion can be derived as both combinations provided appropriate environment in addition to the readily available raw materials. These factors expedited the vermi-composting process (Suthar, 2009). Comparatively, KW although provide readily available material for decomposition, it was less favourable to the worms due to the presence of cooking oil, spices and others. In addition, the absence of unwanted organisms including larvae of insects in VCS2 only allows the worms to fully utilize the food supply. This resulted with time taken for complete conversion to be 28% longer than VCS1. The longest to completely degrade was KW+GC. This is probably due to the presence of cellulosic materials which slowed down the degradation process. However the degradation rate in VCS2 is 25% faster than that of VCS1. It could be due to the competition between worms and unwanted organisms within VCS1 system. The vermi-composting systems were slightly acidic (pH 5.8-7.6) due to the generation of organic acids during the degradation process (Suthar, 2009; Garg *et al*., 2006).

Products from VCS2 were more homogenous with soil like texture (Plate 3.24). This indicated complete conversion of the organic materials into humus (Suthar, 2009; Raut *et al*., 2008). Results from the analysis of VCS2 products are depicted in Table 3.18.

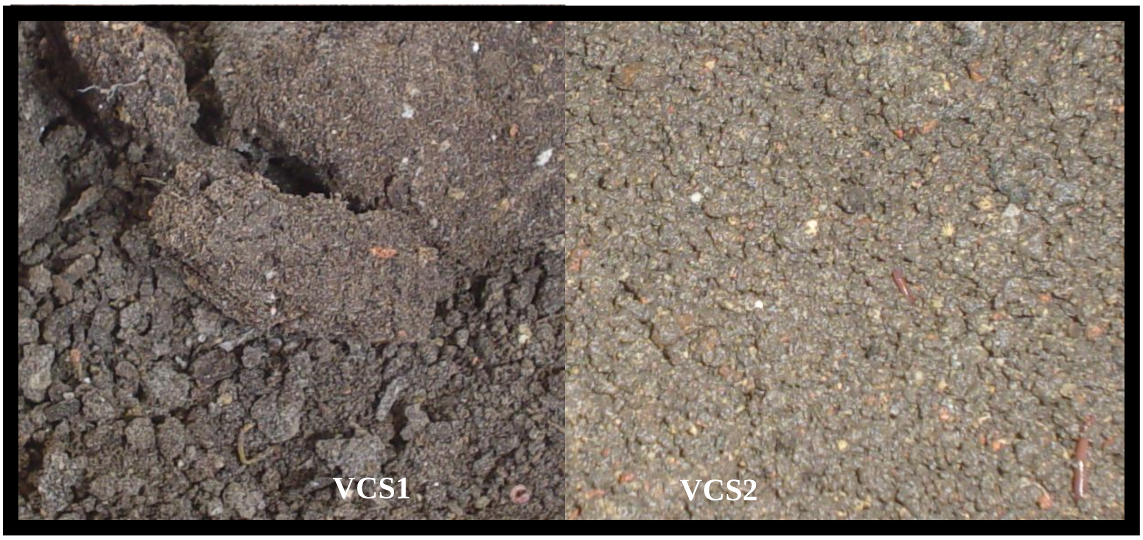


Plate 3.24: Vermi-compost produced in VCS1 and VCS2.

Table 3.18: Analysis of the vermi-compost in VCS2.

Combination	pH	Water holding capacity (%)	Total organic carbon (%)	Nitrate (mg/kg)	Phosphate (mg/kg)	Potassium (mg/kg)
KW	5.8±0.4	77.9±1.1	12.7±1.5	46.5±0.8	21.5±3.1	18.4± 0.3
KW + GC	6.5±0.1	65.2±0.7	13.2±0.2	18.3±0.3	53.3±0.3	9.7± 1.7
KW + GM	7.6±0.1	76.3±2.3	12.4±0.3	26.2±2.4	321.4±12.7	24.8±4.4
KW + VC	6.5±0.2	78.6±3.1	12.2±1.6	27.44±1.0	33.2±1.9	19.1±1.1
KW + GS	7.1±0.1	74.3±1.4	11.1±1.3	6.5±0.7	65.8±2.1	8.8±0.7

Water holding capacity and the total organic carbon of the vermi-composts were improved 12-17% and 9-14% from the initial stage values to 65% to 79% and 11% to 13%, respectively.

Since there is no global or local standards for vermi-compost in Malaysia, standard for compost in EU and USA is used as references. All of the concerned elements such as Cr, Cu and Cd, were below detection limit or within the acceptable range of both EU limit and USA Biosolid limit. Aside from Pb, Cr and Cd, metal elements level increased approximately 32-45% from the initial level of the starting materials. Analysis of the metal elements in the final products is shown in Table 3.19.

Table 3.19: Metal elements in vermi-compost (mg/kg) from VCS2.

Samples	Cr	Cu	Cd	Hg	Ni	Zn	Pb
KW	nd	0.01±0.003	nd	9.5±2.1	10.0±0.5	5.7±1.7	nd
KW+ GC	nd	0.02±0.005	nd	9.6±4.5	10.9±0.7	17.3±2.2	nd
KW+ GM	nd	0.01±0.008	nd	8.6±1.1	9.5±2.5	16.9±0.8	nd
KW+ VC	nd	0.04±0.001	nd	3.3±0.7	7.3±1.7	42.9±1.5	nd
KW+ GS	nd	0.02±0.007	nd	8.9±1.3	10.7±0.3	22.1±3.1	nd
EU limit range*	70-200	70-600	0.7-10	0.7-10	20-200	70-1000	210-4000
USA biosolid limit*	1200	1500	39	17	420	300	2800

Note: nd = not detected; * (Gupta and Garg, 2008)

The concentration of Hg in VCS2 was at the high end of the limit allowed. This probably resulted from the presence of Hg in the food source such as rice. Rice was reported to contain certain level of Hg (Extoxnet, 1990). Though no regular pattern is observed in final vermi-composts in relation to initial concentration, the increase in heavy metal content could probably be attributed to the massive degradation process of the raw materials and the mineralization process. This was supported by findings on various types of raw materials used in vermi-composting (Suthar, 2009; Yadav and Garg, 2009; Adi and Noor, 2009; Plaza *et al.*, 2008; Elvira *et al.*, 1998).

Therefore, vermi-composting with VCS2 can be considered as an option to convert household organic waste to value added product and reduce its disposal into landfills. Other alternative route is the conversion to refuse-derived fuel (RDF).

3.3.4.2 Refuse-derived fuel conversion

The average calorific value of the combustible waste generated by Malaysians is 23,000 kJ/kg. It is comparable with findings of calorific value of MSW generated by other countries which ranged from 19,000 kJ/kg to 21,000 kJ/kg (Hansen *et al*., 2007). Plastic generates the highest calorific value at 44,000 kJ/kg, followed by textile and wood at 21,000 kJ/kg and 19,000 kJ/kg, respectively. Figure 3.13 illustrates the average calorific value of different types of waste from the study.

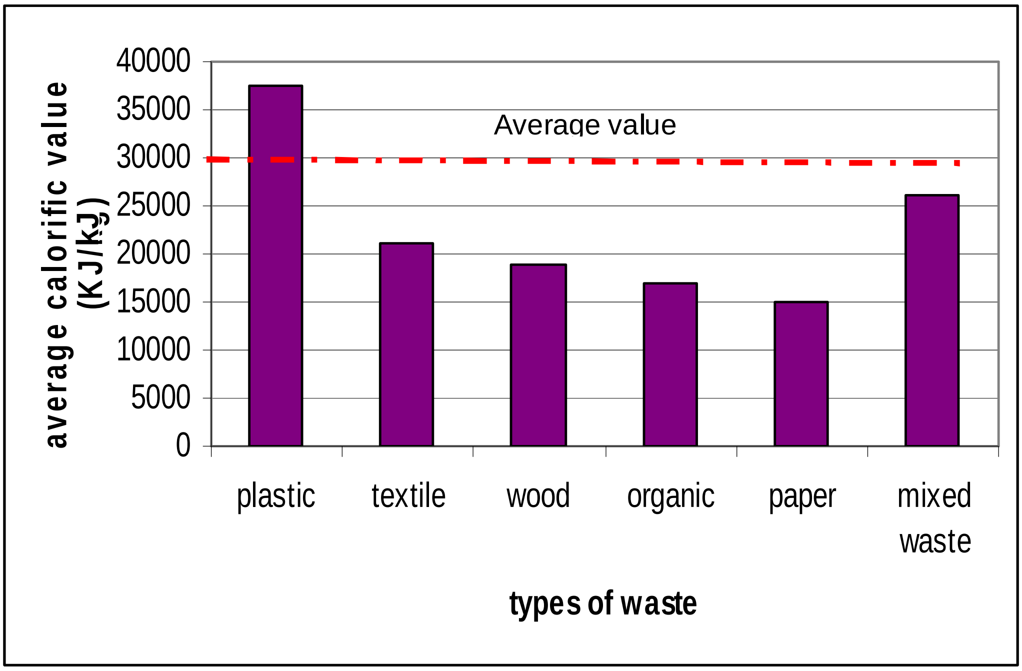


Figure 3.13: Average calorific value for different waste types.

Mixed wastes consisting of plastic, rubber and/or textile has a calorific value of 26,000 kJ/kg. The value is lower than that of plastic samples but higher than those of textile, wood and others. Rubber waste was reported to exhibit high calorific value which enabled the material to be converted into liquid fuel (Huang and Tang, 2007). The calorific value of organic matter was 22% lower than the average due to the high moisture content it retained. This is agreeable with previous findings (Hansen *et al*., 2007; Kathirvale *et al*., 2003; Lucas *et al*., 2000). Figure 3.14 depicts the different calorific values generated by the various types of plastic waste.

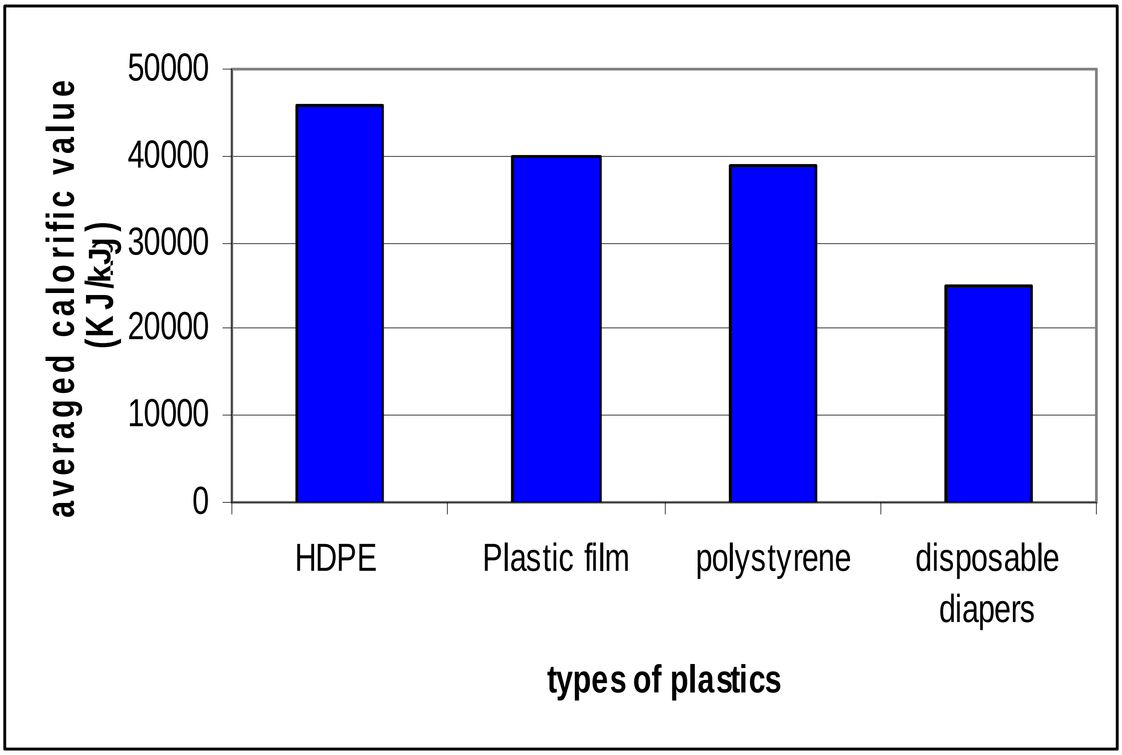


Figure 3.14: Average calorific value for different plastic wastes.

The complexity of the polymeric chain is the main factor influencing the calorific value of the plastic waste samples. The more complex and denser the polymeric chain the higher is the calorific value (Lotfi, 1997; Keuton and Wilmer, 1992). This resulted with high density polyethylene (HDPE) samples generated the highest calorific value (46,000 kJ/kg) followed by plastic film (40,000 kJ/kg) and polystyrene (39,000 kJ/kg). Even though disposable diapers consist of approximately 23% polypropylene plastics and 43% wood pulp, the remaining 27% consist of super absorbent gel polymer retains high moisture content (Popescu *et al*., 2008). This resulted with the low calorific value (25,000 kJ/kg) among the plastic types. Moisture level reduces the burning efficiency and lowered the calorific value (Magrinho and Semiao, 2008; Liu *et al*., 2008; Kathirvale *et al*., 2003).

Between plastic bags, the non- degradable types has higher calorific value (43,000kJ/kg) than the degradable bags (37,000kJ/kg). The fact that degradable bags had undergone series of alteration with various additives resulted with the polymeric chain becoming weak for degrading process. This resulted with approximately 14% less calorific value of degradable bags than that of the conventional types. Studies also indicated that packaging waste such as tetrapaks has an averaged calorific value of 22,000 kJ/kg. This is lower than the value displayed by other plastic wastes which probably due to the presence of non-combustible layers. In addition, the trilaminar structure of the material with the capacity to trap moisture could also lower its calorific value.

Newspaper has the highest calorific value (17,000 kJ/kg) (Figure 3.15) probably contributed by the factor that newspaper production involved the usage of some percentage of recycled paper. The repetitions in newspaper recycling synergistically increased the calorific value by making it more thermally stable and firmly attached with the ink (Filho *et al*., 2008; Zhang *et al*., 2008). Studies by Conesa *et al*. (2008) indicated that even the residual material of paper recycling process contained approximately 94,000 kJ/kg calorific value due to the de-inking process which contributed to higher calorific value of the recycled paper. Similarly with the production of corrugated paper where certain percentages of recycled corrugated paper is utilized in the production of corrugated paper resulting with more compacted structure that increases the calorific value (16,000 kJ/kg). On the other hand, white paper which contains very high percentage of virgin fibrous materials resulted with lower calorific value (14,000 kJ/kg) than those which contain less percentage of virgin materials.

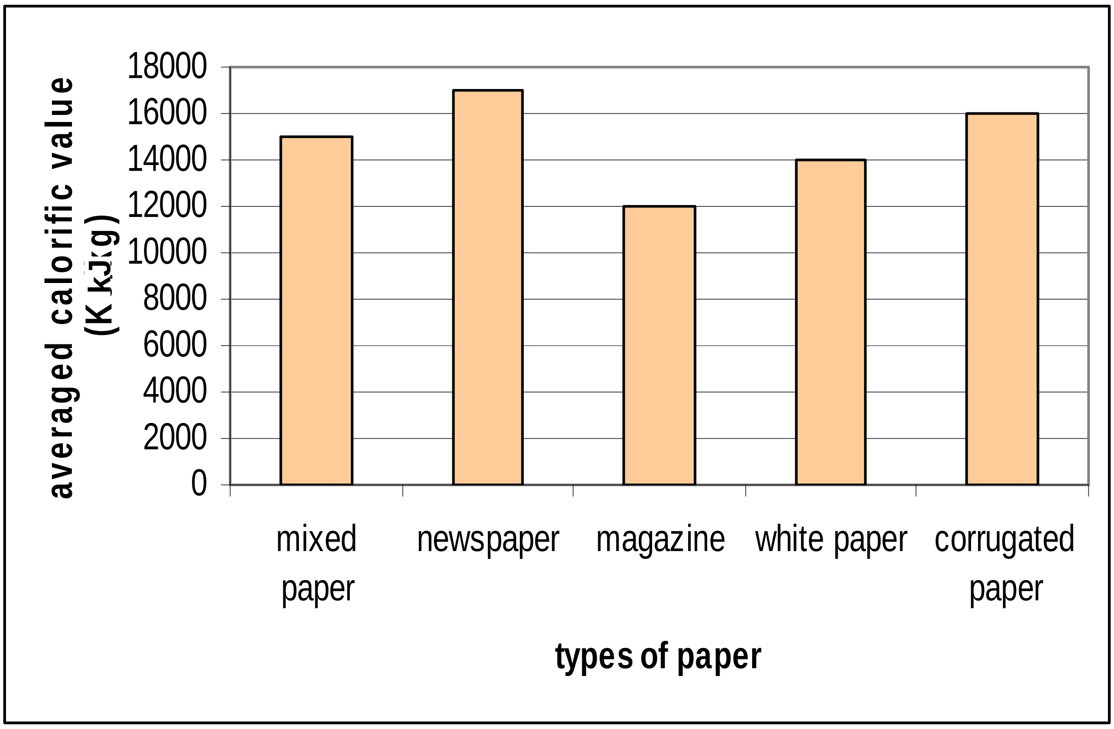


Figure 3.15: Average calorific value for different types of paper wastes.

Lack of the reprocessing procedure resulted with low density and compaction (Magrinho and Semiao, 2008; Li and Liu, 2000). Mixed paper has approximately 7% higher calorific value than white paper probably due to the presence of various combinations of recycled and new paper materials. Magazine samples have an average calorific value of 12,000 kJ/kg making it the lowest in the paper waste samples. This is probably due to the multilayer chemical coating used to maintain the glossy feature of the magazine surface (Lotfi, 1997). The process may result with high percentage of trapped moisture content between the layers which reduces the calorific value of the materials.

Textile waste has a calorific value of 21,000 kJ/kg which is the third highest among the different types of combustible waste. Detailed study indicated that coloured textile samples have 10% higher calorific value than the non-coloured textile. This probably contributed by the complex bonding of the additional colouring compounds within the textile material resulting with more energy being released during combustion.

Wood materials have an averaged calorific value of 19,000 kJ/kg. Multi-layer and multi-coating wood samples have approximately 6% higher calorific value than the non-layering and non-coating types. This is probably due to the presence of chemical compound which act as the adhesive between the layering and coating. The additives improved the dimensional stability and strength properties of the composites (Adhikary *et al.*, 2008) which increase the total calorific value of the materials. However, the calorific value of the wood samples are 10- 22% lower than that of textile and plastic. This is probably due to wood’s high water holding capacity to absorb and retain moisture within the structure that the calorific value reduces. The combustion of wood is normally partial due to the complex cellulose structure which trapped moisture particles. This would cause the calorific value to reduce accordingly.

Organic waste which consists of kitchen waste and garden waste has an average calorific value of 18,000 kJ/kg. The calorific value was reduced due to its high moisture content which ranged 40-50% in most organic waste (Magrinho and Semiao, 2008). Calorific value of the kitchen waste was approximately 6% lower than that of the garden waste. This is contributed by the fact that Malaysian kitchen waste contains high moisture resulting with more energy is needed to fully combust the materials (Fauziah and Agamuthu, 2004).

Due to the high calorific value of certain waste material, conversion to RDF will allow the diversion of waste from disposal sites. This is applicable with the implementation of appropriate waste sorting system.

3.3.5 Leachate Analysis

Detailed leachate characteristics are depicted in Appendix 3.19. Results indicated that values of the parameters differed from one landfill to the others. BOD ranged from, as low as, 12.5 mg/L to, as high as, 625 mg/L, while COD was 169 mg/L to 9029 mg/L. The high differences of the leachate characteristics were mainly due to the different waste components disposed into the landfills (Sormunen *et al.*, 2008; Visvanathan *et al.*, 2007; Fauziah *et al.*, 2005; Keenan *et al.*, 1984). In addition, different practices in waste compaction, landfill cover material and other factors also affect the quality and quantity of leachate generation (Heyer *et al.*, 2005; Blight, 2005; Őzkaya *et al.*, 2005). The pH value ranged from pH6.4 to pH8.1. This indicates that the landfill conditions are either moving from acidogenic phase to methanogenic phase or currently undergoing methanogenic phase. Landfill is undergoing methanogenic stage when pH of leachate ranged from pH7.5-9.0 (Jun *et al.*, 2009; Berthe *et al.*, 2008; Kruempelbeck and Ehrig, 1999). Conductivity of the leachate was found to differ from one landfill to the others. It is due to the different age of landfills involved in the studies. Conductivity and the activity of leachate generated was reported to be directly proportional to the age of the landfill (Ziyang *et al.*, 2009; Jun *et al.*, 2009; Sormunen *et al.*, 2008). Table 3.20 indicates the average value of the leachate from the nine landfills in Selangor and the standard deviations. COD averaged at 4313 mg/L while BOD was 194 mg/L. Correlation for sulfite and chloride is the most significant with a linear regression (R^2= 0.8786) than that of logarithm (R^2=0.6463) and exponential regression (R^2=0.6185). The positive correlation between sulfite and chloride (ρ= 0.94) is depicted in Figure 3.16.

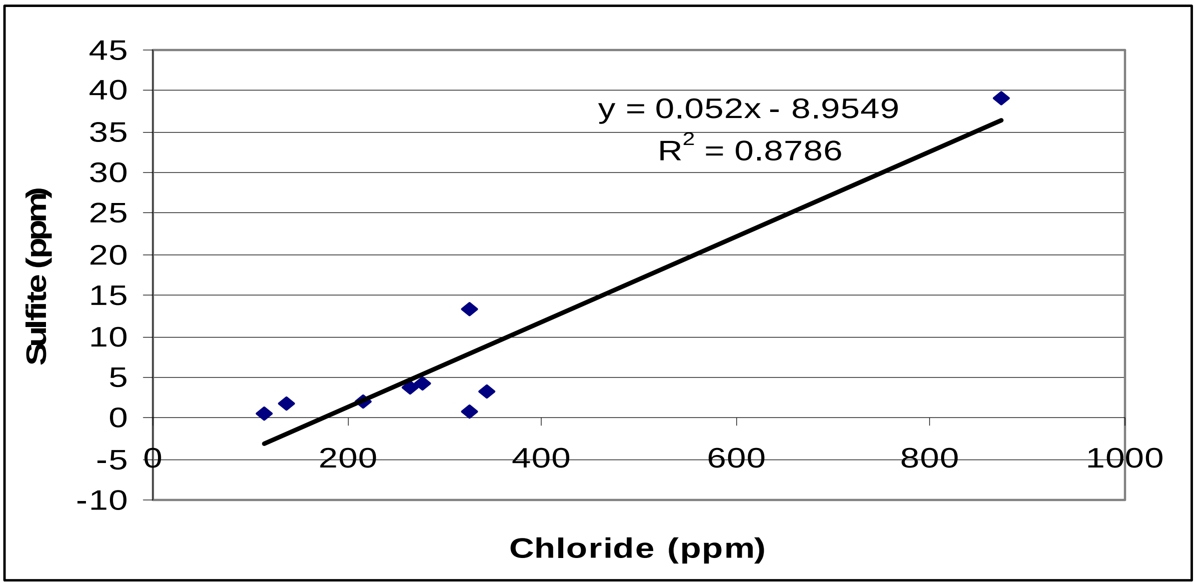


Figure 3.16: Concentration of sulfite versus chloride in leachate samples.

Table 3.20: Average values of leachate from nine landfills in Selangor.

Parameter	Concentration	Standard B (EQA 1974)
pH	7.38 ± 0.63	pH 5.5-9.0
BOD_5 (mg/L)	194.93 ±128.16	50
COD (mg/L)	4,313.71 ±2933.72	100
TOC (mg/L)	2,441.46 ±1585.9	-
Total Nitrogen (ppm)	43.09 ±28.36	-
Turbidity (NTU)	202.03 ±161.01	-
Conductivity (μs)	175.24 ± 106.82	-
Hardness	106,579 ±84,432.63	-
Salinity (ppm $CaCO_3$)	2.25 ±1.33	-
Alkalinity	340.22 ±313.56	-
Chloride (ppm)	319.56 ±222.79	-
Sulfite (ppm)	12.37 ±7.66	-
Iron (ppm)	4.18 ±2.06	-
Colour (PtCo)	1,265.82 ±1123.99	-
Total Phosphorous (ppm)	146 ±14.0	-
Total Solid (mg/L^{-1})	8.19 ±5.58	-
TSS (mg/L^{-1})	0.38 ±0.5	100

Both chloride and sulfite concentrations increase with the increase in the mineral concentration in the leachate (Renou *et al*., 2008). The concentration of chloride was also found to have linear regression with Fe (R^2= 0.52) as indicated in Figure 3.17 with correlation coefficient ρ=0.72.

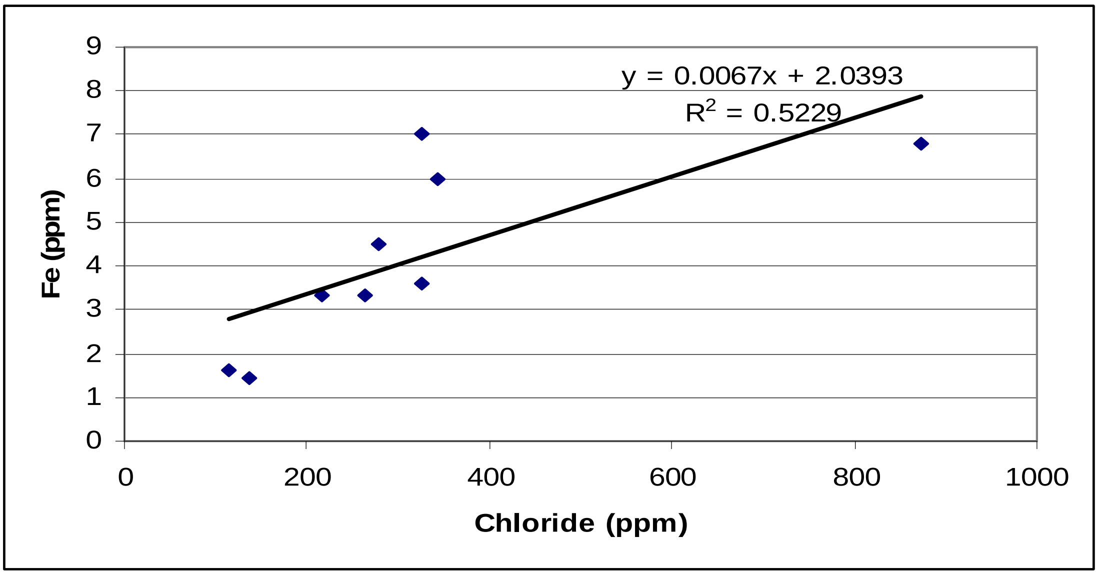


Figure 3.17: The regression of Fe and chloride concentrations in leachate.

The analysis of leachate from the nine landfills indicated that increase in chloride concentration will result with direct increase in Fe concentration. Increase in Fe was also found to have a correlation with turbidity of the leachate. A logarithmic regression (R^2= 0.7729) between Fe concentration and leachate turbidity was derived with correlation coefficient ρ= 0.70 as shown in Figure 3.18.

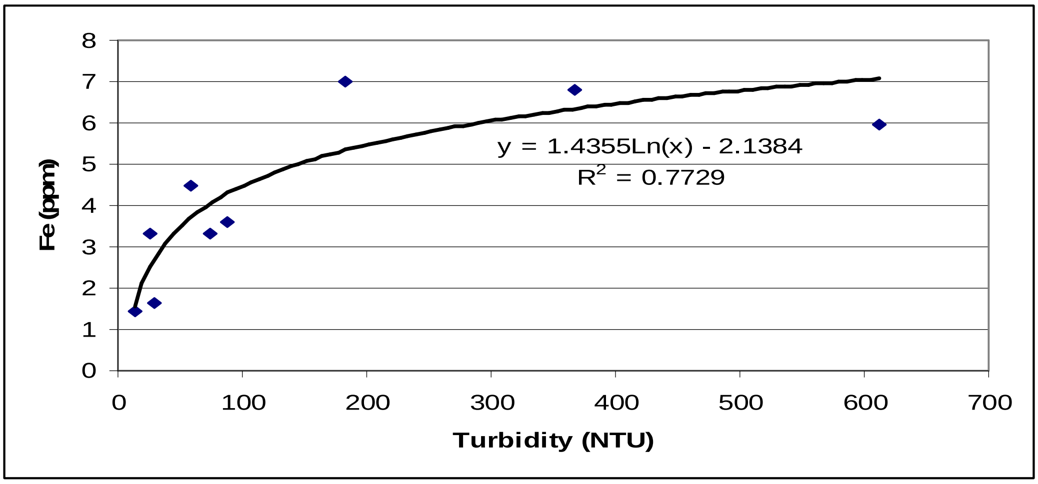


Figure 3.18: Logarithmic regression between Fe concentration and the turbidity.

The turbidity of leachate increases with the presence of high concentration of Fe as Fe will be oxidized and turned the water darker. It was observed that significant correlations (ρ> 0.50) exist between the parameters. Most of them are positive correlations indicating direct increase in one parameter will result with the increase in another. However, two negative correlations were obtained between salinity and conductivity, and salinity and total nitrogen. These negative correlations imply the increase value of one parameter will result with the decrease value of another. Other significant correlations derived from the analysis of leachate from nine landfills are summarized in Table 3.21.

Table 3.21: Correlation coefficient of various parameters in leachate analysis.

Parameters		**Correlation coefficient (ρ)**	**Coefficient of determination (R^2)**
Array 1	**Array 2**		
pH	Chloride	0.51	IS
	Sulfite	0.58	IS
	Fe	0.72	0.51 (Appendix 3.20)
	Colour	0.82	0.67 (Appendix 3.21)
Salinity	Conductivity	-0.50	IS
	Total nitrogen	-0.72	0.52 (Appendix 3.22)
Alkalinity	Sulfite	0.56	IS
	Fe	0.67	IS
Chloride	Turbidity	0.60	IS
	Colour	0.74	0.54 (Appendix 3.23)
	BOD	0.92	0.85 (Appendix 3.24)
	Total solid	0.89	0.79 (Appendix 3.25)
Sulfite	Fe	0.66	0.65(Appendix 3.26)
	Colour	0.77	0.60(Appendix 3.27)
	BOD	0.92	0.85(Appendix 3.28)
	Total solid	0.89	0.86(Appendix 3.29)
Iron	Turbidity	0.66	IS
TSS	Hardness	0.53	IS
	COD	0.61	IS
Colour	BOD	0.65	IS
	Total solid	0.72	0.52(Appendix 3.30)
BOD	Total solid	0.96	0.92(Appendix 3.31)

IS = insignificant linear regression (R^2< 0.50)

Since most of the landfills are without proper leachate collection pond (Plate 3.25), the concentrations of metal elements in the leachate samples fluctuate from one landfill to the others depending on the surface water received by the landfills. Metal elements detected in the leachate were below the Standard B EQA 1974 with the exception to Cr (145 ppm). This is due to the indiscriminate disposal of Cr-containing components such as stainless steels based waste, paints and tanning products. The average of Ca and Mg were also high due to its presence in putrescible waste which accounted the highest portion (>50%) of the waste received by the landfills in Selangor (Fauziah and Agamuthu, 2004). Table 3.22 depicts the average concentration of metal elements from the nine landfills.

Table 3.22: Average concentration of metal elements in leachate from nine landfills in Selangor.

Elements	Concentration (µg/l)	Standard B (EQA 1974) (µg/l)
Cu	85.3 ± 0.71	1000
Zn	231.2 ± 3.4	1000
Ni	107.9 ± 4.2	1000
Co	23.4 ± 0.92	-
Mn	225.4 ± 1.9	1000
Cr	**144.5** ± 2.3	50
Ca	52 432 ± 114.5	-
Mg	18 675 ± 124.5	-
Cd	1.548 ± 0.251	20
Ba	62.4 ± 1.46	-
Pb	41.2. ± 2.68	500
As	1.83 ±1.06	100
Al	374.4 ± 10.3	-
V	19.1 ± 0.94	-
Se	7.98 ± 2.5	-
Ag	4.74 ± 0.78	-
Sr	137.4 ± 4.5	-
Li	49.8 ± 1.42	-

Plate 3.25: Some of the leachate collection points in Kundang disposal site.

With exception of Cr, all pollutants concentration were below the Standard B EQA 1974. The leachate has low pollutant concentration mainly due to surface water dilution. However, the treatment of leachate is still necessary if the accumulation effects are taken into consideration. Daily pollutant loading from leachate flowing from Kundang landfill was calculated and it was found to have high risks in contaminating the adjacent river (Fauziah and Agamuthu, 2005).

3.3.6 Leachate Treatment System

3.3.6.1 Physico-chemical treatment using ferric chloride

Results from the physico-chemical treatment of the leachate indicated that parameters such as BOD_5, COD, colour, turbidity and others were reduced with the addition of $FeCl_3$. On the other hand, addition of alum was found to be less effective (Plate 3.26). Consecutive sections discuss the results obtained from the coagulation and flocculation process of leachate with $FeCl_3$ at different concentrations and different pH. Results obtained from the utilization of different concentration of $FeCl_3$ are detailed in Appendix 3.32. Reductions of BOD (ρ= -0.94) and COD (ρ= -0.92) ranged from 7% to 12% and 0.4% to 7%, respectively, at different $FeCl_3$ concentration. Figure 3.19 illustrates BOD and COD removal with increase in $FeCl_3$ concentration.

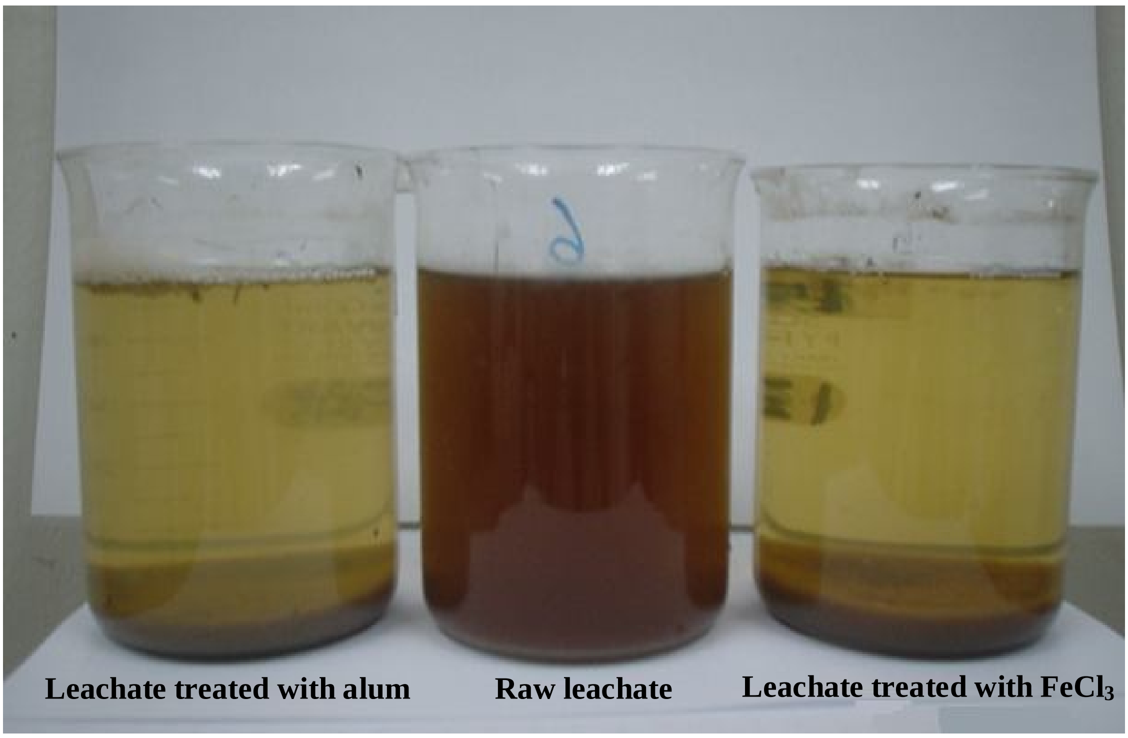


Plate 3.26: Raw and treated leachates.

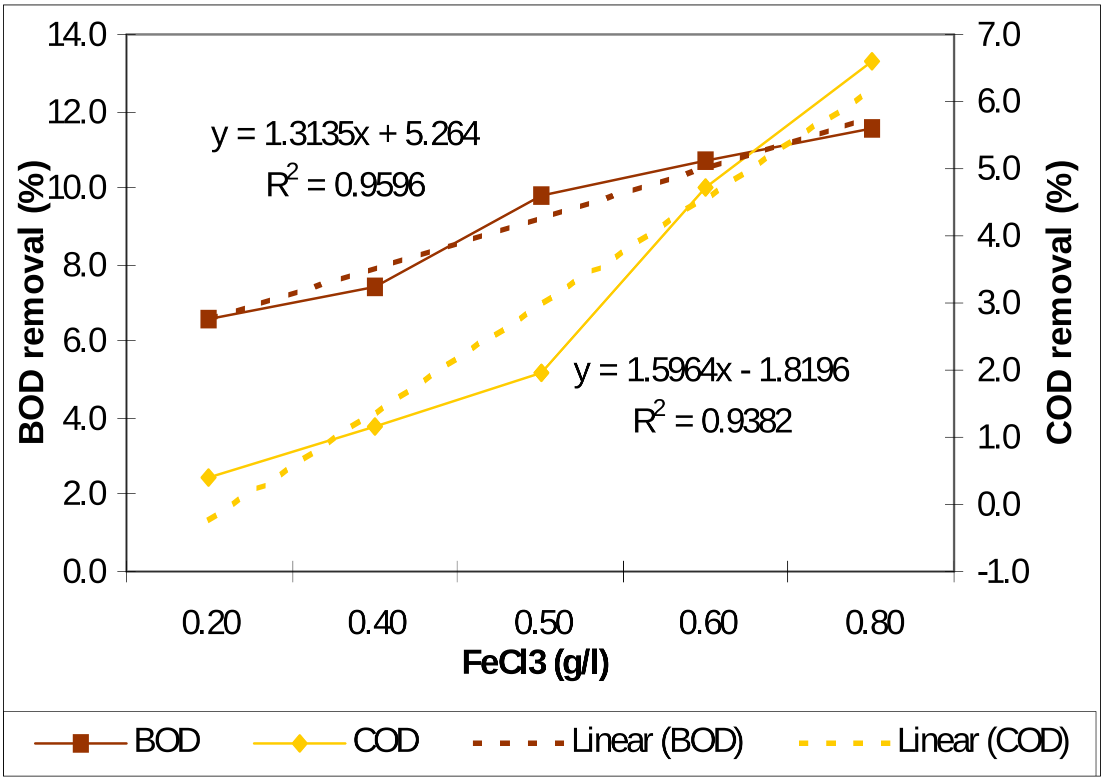


Figure 3.19: Percentage of BOD and COD removal at different $FeCl_3$ concentrations.

Significant linear regressions were plotted between BOD and COD reductions, and $FeCl_3$ concentrations. The ability to remove more organic and non-organic compounds increased at higher concentrations of $FeCl_3$. It is agreeable with the leachate physico-chemical treatment in many reports (Kang *et al*., 2002; Wang *et al*., 2009). Significant correlation (ρ= 0.78) was also derived between the reduction in BOD and COD in the treatment indicating a linear regression between the two parameters as depicted in Figure 3.20.

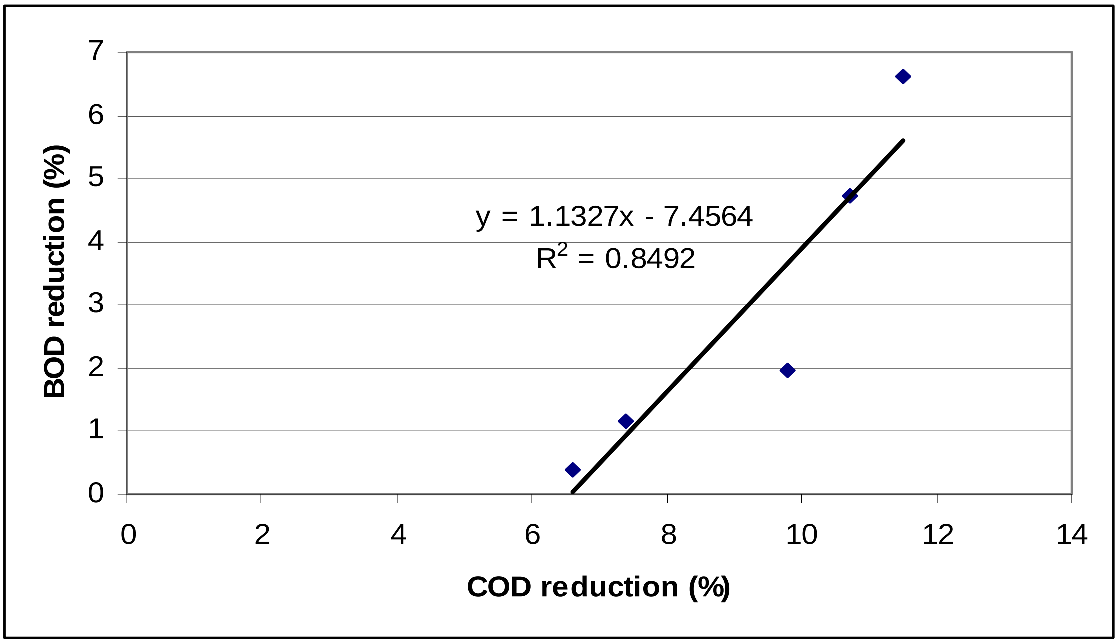


Figure 3.20: Linear regression between BOD and COD reduction in the treatment system.

The final pH (ρ= 0.99) and alkalinity (ρ= 0.93) of the treatment system is correlated with $FeCl_3$ concentrations (Appendix 3.33). The treatment system became less basic with increase in $FeCl_3$ concentration. The final pH was also found to have significant correlation with colour (R^2= 0.8515) and turbidity (R^2= 0.7444) (Appendix 3.34). In addition, colour (ρ= -0.99) and turbidity (ρ= -0.92) were significantly correlated with the concentration. The increase in concentration of $FeCl_3$ resulted with lower intensity in both parameters illustrated in Figure 3.21. Coloidial matters that present in leachate which affect the level of colour and turbidity, react better at higher concentration. This increased the reduction of colour and turbidity (Primo *et al*., 2008).

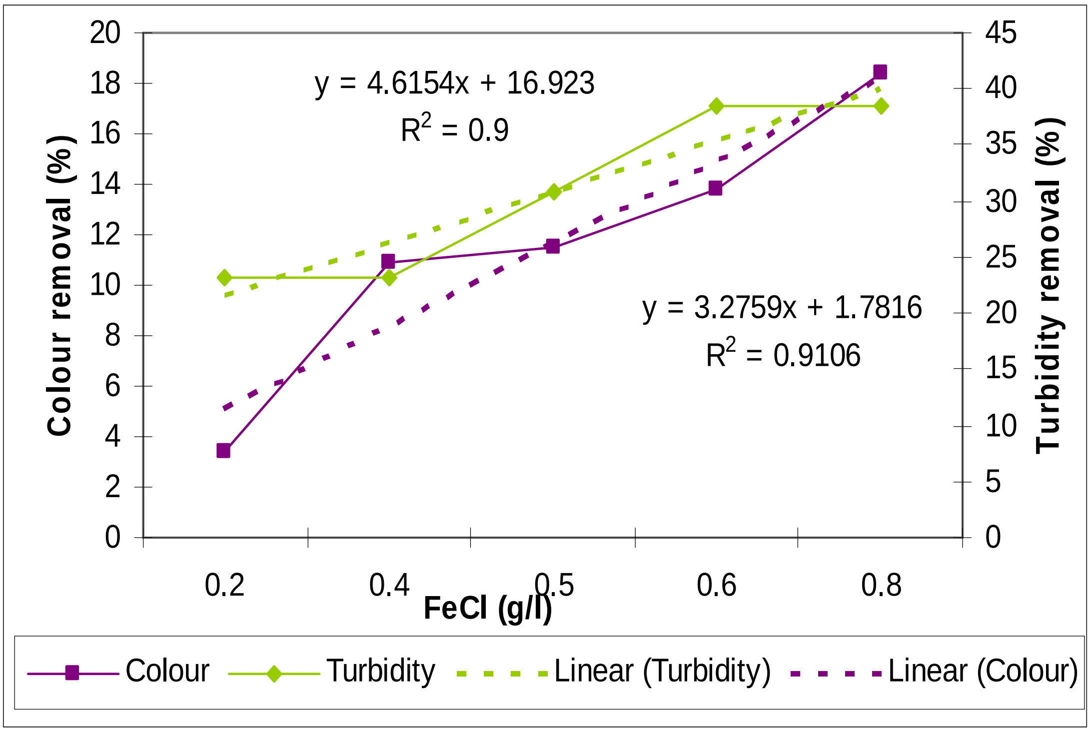


Figure 3.21: Linear trends in colour and turbidity removal with $FeCl_3$ concentrations.

The removal efficiency of Total Solid (TS) and Total Dissolved Solid (TDS) ranged 17-49% and 5-22%, respectively. TS (ρ= -0.94) and TDS (ρ= -0.93) indicated decreasing trends with increasing $FeCl_3$ concentrations (Figure 3.22). This probably is resulted by the increase activity of the coagulant to bind solid particles within the treatment system to allow them to settle (Zhang and Wang, 2009).

The removal efficiency of ammonical-N ranged between 11 to 50% while total P removal was 29 to 69%. It was significantly correlated indicating higher removal efficiency at higher $FeCl_3$ concentrations. Similar findings have also been reported in treatment of landfill leachate from China and northern Spain (Zhang and Wang, 2009; Primo *et al*., 2008). The trend in ammonical-N and total P removal is illustrated in Figure 3.23.

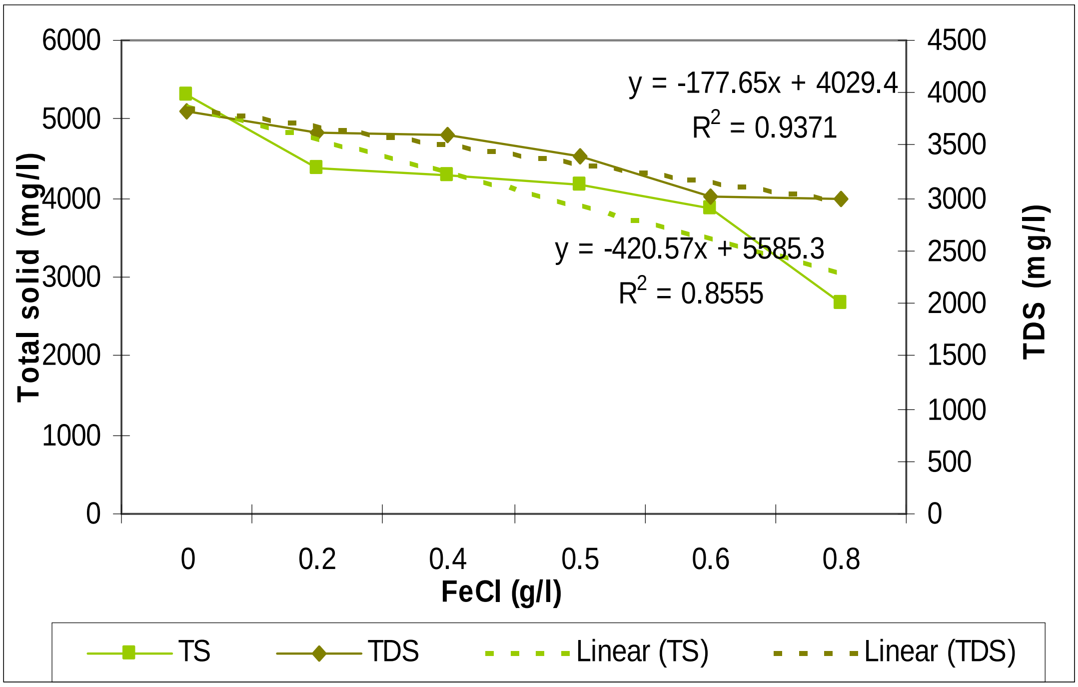


Figure 3.22: TS and TDS value at different concentration of $FeCl_3$.

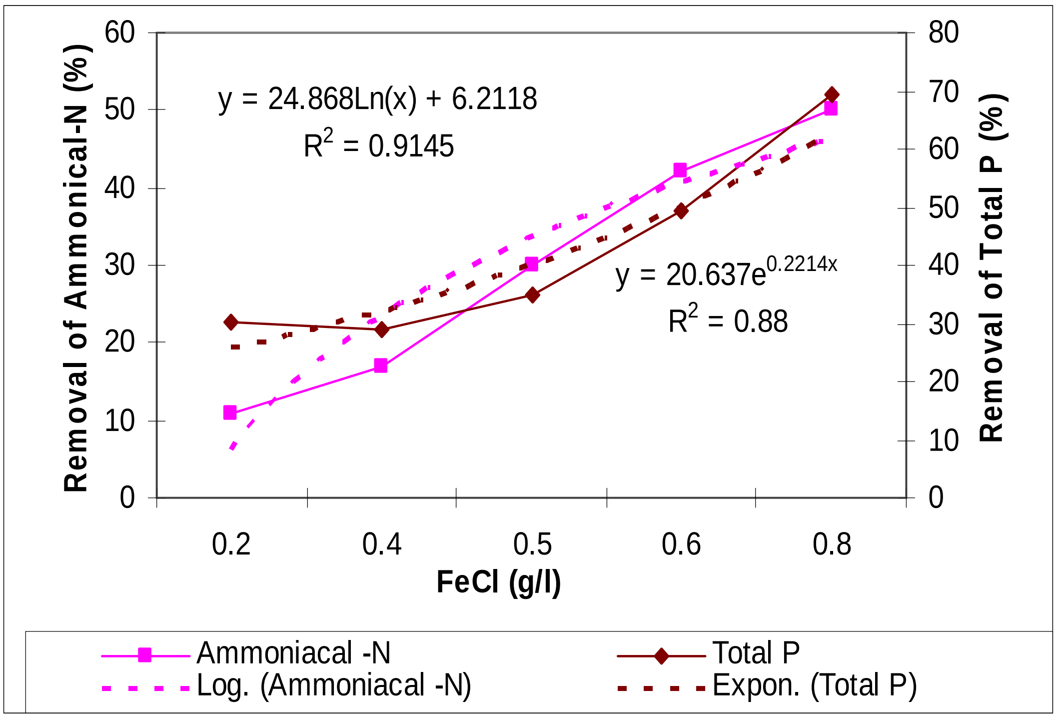


Figure 3.23: Removal in ammonical-N and total P by different concentration of $FeCl_3$.

The removal of Cr and Co also showed a similar increasing trend (Figure 3.24) as $FeCl_3$ concentrations increased with correlation coefficient ρ= -0.92 and ρ= -0.94, respectively.

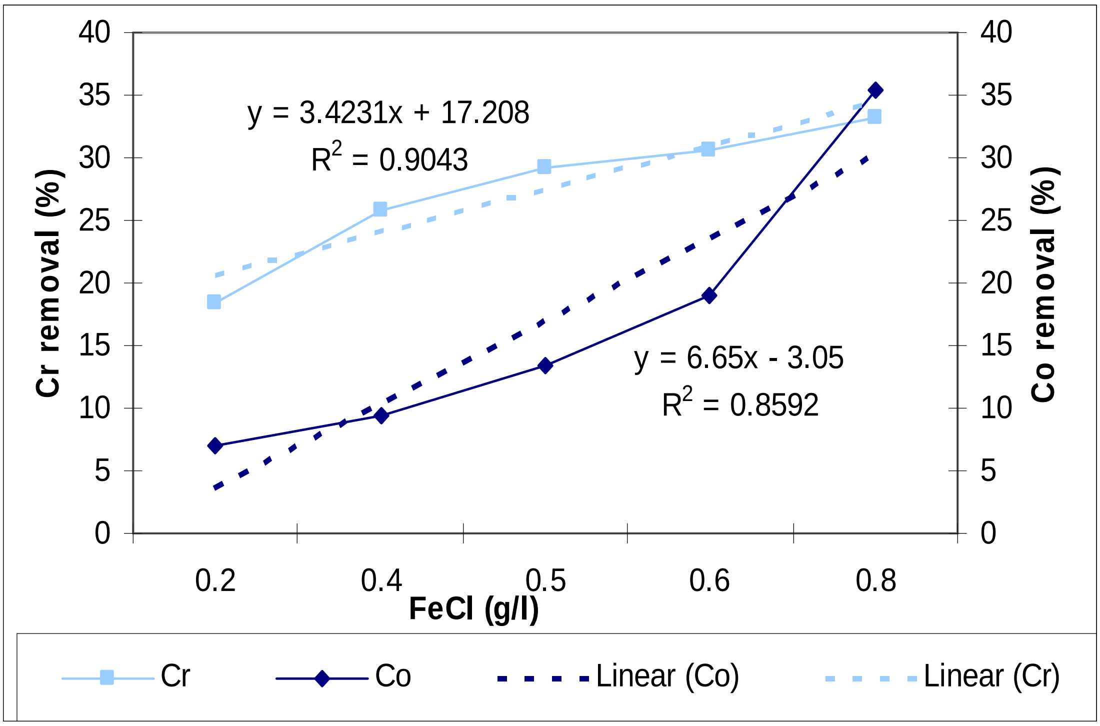


Figure 3.24: Removal of Cr and Co at different $FeCl_3$ concentrations.

Removal efficiency of Cr and Co ranged at 18 to 33% and 7 to 36%, respectively in a linear trend. Removal of other metal elements were also found to have significant correlation with the $FeCl_3$ dosages. Table 3.23 summarizes the removal efficiency, correlation coefficient and the regression (coefficient of determination) between the analyzed parameters and the concentration of $FeCl_3$ (0-0.8 mg/L).

Table 3.23: Summary of the removal efficiency, correlation coefficient and coefficient of determination for different parameters at different concentration of $FeCl_3$.

Parameters	Removal (%)		Correlation coefficient (ρ)	Coefficient of determination (R^2)
	Min	Max		
BOD	6.6	11.5	-0.938	0.9596 (Linear)
COD	0.4	6.6	-0.920	0.9382 (Linear)
TS	17.4	49.4	-0.941	0.7844 (Linear)
pH	0.6	1.8	-0.990	0.9655 (Linear) Appendix 3.32
TDS	5.1	21.8	-0.933	0.8179 (Linear)
Salinity	0	22.2	-0.931	0.9386 (Linear)
Conductivity	4.5	22.4	-0.971	0.9258 (Linear)
Total P	28.8	69.5	-0.955	0.88 (Exponential)
Hardness	4.8	28.6	-0.913	0.6856 (Linear)
Alkalinity	18.8	36.8	-0.925	0.7656 (Linear) Appendix 3.32
Sulfite	16.7	33.3	-0.836	insignificant
Colour	3.4	18.4	-0.991	0.9 (Linear)
Turbidity	23.1	38.5	-0.927	0.9106 (Linear)
TSS	12.5	12.5	-0.714	insignificant
Ammonical-N	10.8	50.2	-0.978	0.9145 (logarithmic)
Cu	18.5	82.0	-0.989	0.9509 (Linear) Appendix 3.36
Zn	17.4	43.8	-0.983	0.9473 (Linear) Appendix 3.36
Ni	5.1	32.6	-0.60	0.8744(Exponential) Appendix 3.35
Co	7.0	35.5	-0.934	0.8592 (Linear) Appendix 3.37
Mn	3.6	38.9	-0.984	0.9955 (Linear)
Cr	18.5	33.2	-0.928	0.9043 (Linear) Appendix 3.37
Ca	14.2	47.6	-0.979	0.922 (Linear)
Mg	7.3	36.2	-0.870	0.7162 (Linear)
Cd	23.8	73.4	-0.879	0.8661 (Linear)
Ba	37.6	77.9	-0.886	0.5889 (Linear)
Pb	3.3	44.0	-0.862	0.5751 (Linear)
Al	12.6	41.2	-0.988	0.9943 (logarithmic) Appendix 3.35
Sr	92	95.1	-0.729	insignificant
Li	1.2	34.7	-0.750	0.6252 (Linear)

The pollutant removal efficiencies have significant correlation with the concentration of $FeCl_3$. Final pH of the leachate after physico-chemical treatments were slightly reduced (0.6-1.8%) from its intial pH. This parameter was found to play role in the efficiency of the coagulation- flocculation process (Primo *et al.*, 2008; Wang *et al.*, 2002). Results from $FeCl_3$ coagulation- flocculation process at different pH is detailed in Appendix 3.38. The removal efficiencies ranged from 2% to100%. BOD and COD were reduced 8 - 12% and 15 - 61%, respectively. With the addition of $FeCl_3$, the more acidic the aqueous system the higher is the reduction efficiency (Figure 3.25).

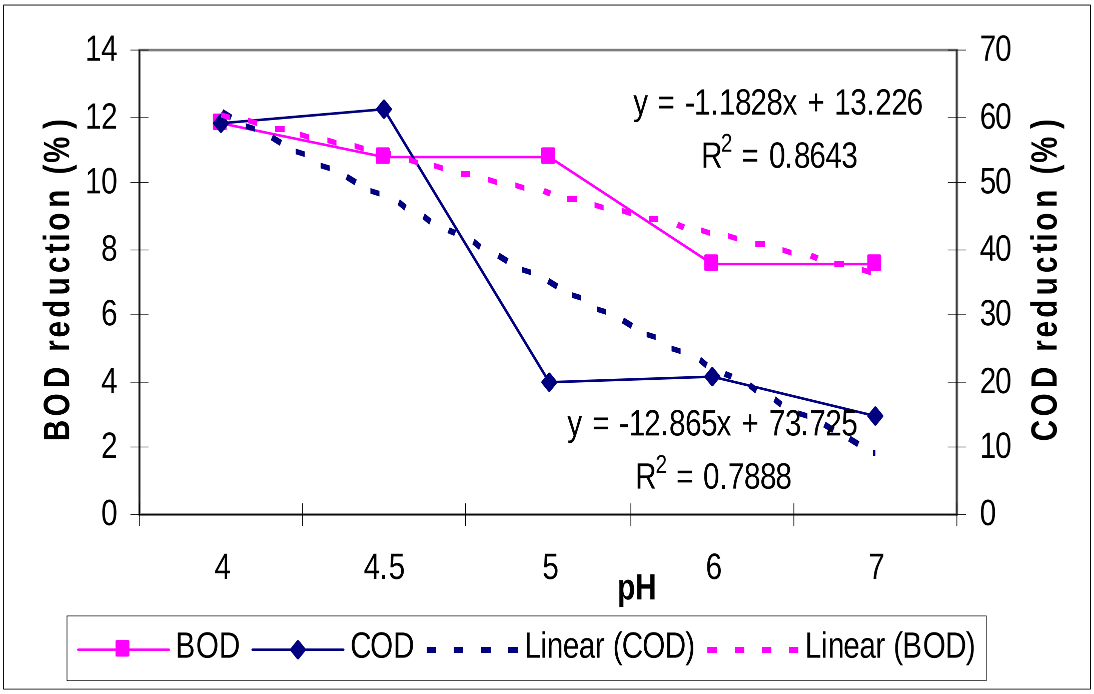


Figure 3.25: Linear regression of BOD and COD removal with $FeCl_3$ at different pH

The more acidic the system the higher is the ability for $FeCl_3$ to bind with the organic matters and remove it from the leachate. In addition, acidic environment also resulted with the digestion of the organic matter which indirectly reduces its intensity at the final analysis. BOD and COD removal displayed a linear trend where removal efficiency decreased when pH moves from acidic to neutral. The correlation of pH and removal efficiency of colour (ρ= 0.82) and turbidity (ρ=0.76) were obtained. The linear trend is depicted in Figure 3.26 indicating more acidic system resulted with higher removal efficiency. Removal of Cd is more efficient in the acidic system where results indicate that Cd reduction has a linear trend with pH (Figure 3.27). In an acidic system, more ions will be formed that is more bonding between Fe^{3+} and Cl^{-}, and the metal elements including Cd formed flocs (Faust and Aly, 1999). This resulted with lower quantity of elements dissolved in the leachate system.

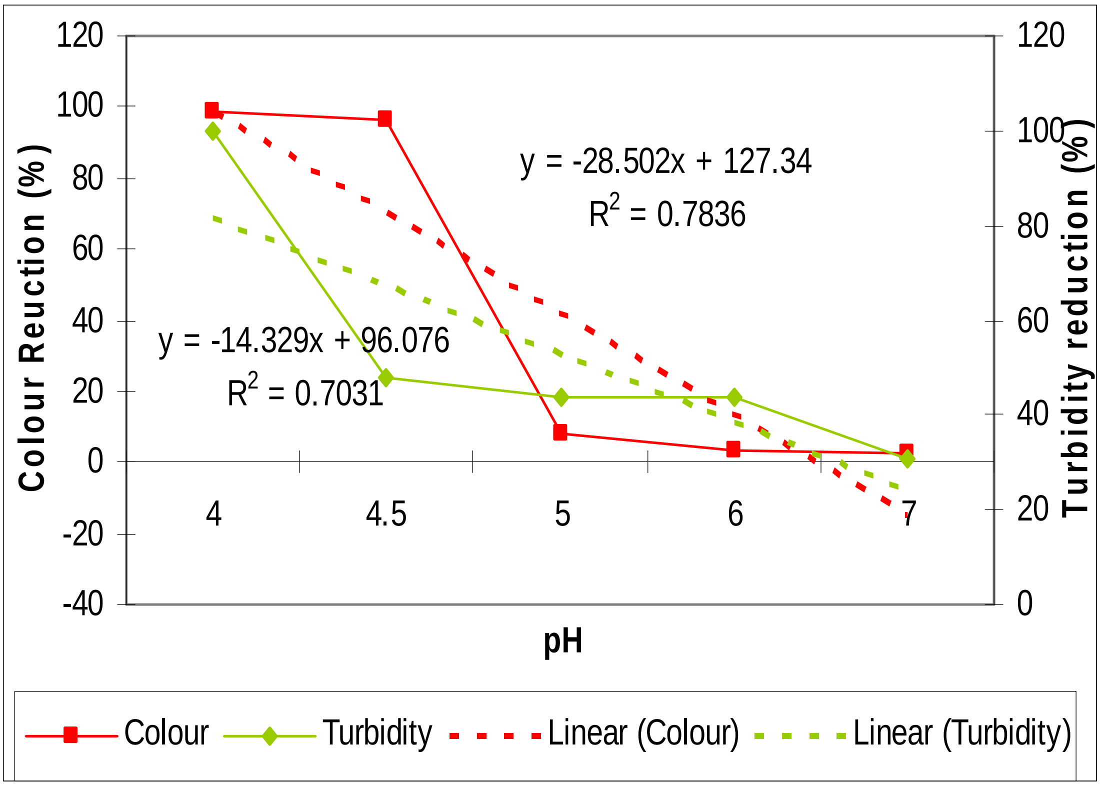


Figure 3.26: Linear trends in colour and turbidity removal by $FeCl_3$ at different pH

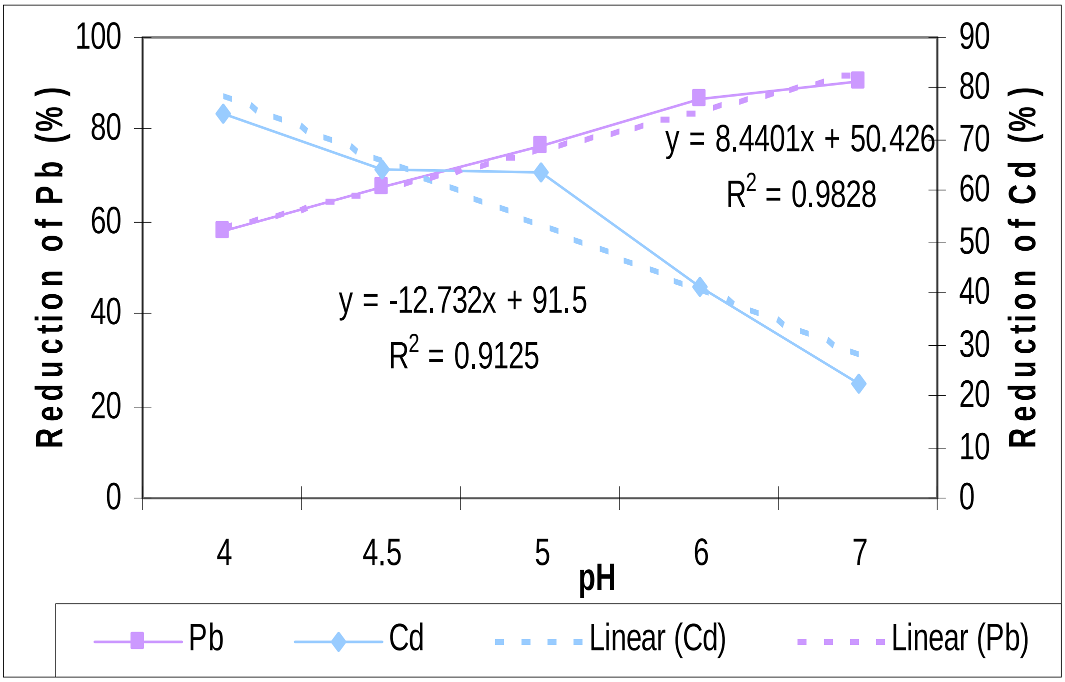


Figure 3.27: Removal efficiency of Pb and Cd by $FeCl_3$ at different pH.

On the other hand, the removal efficiency of Pb has an opposite effect with pH where the more acidic the system the lower is the removal efficiency (ρ= -0.96) (Figure 3.27). This phenomenon probably is because Pb and Ba whose high atomic weight has lesser bonding affinity with other ions in the presence of high concentration of hydroxide ions. This is due to competition of hydroxide ion with the organic compounds for metal adsorption sites to form flocs (Zhang and Wang, 2009). The majority of other metal elements display a positive linear trend (ρ> 0.50) where more acidic the system the higher is the removal efficiency. Table 3.24 summarizes the removal efficiency, correlation coefficient and the regression (coefficient of determination) between the analyzed parameters and the pH of $FeCl_3$ system (pH3.0-pH 8.0).

Table 3.24: Summary of the removal efficiency, correlation coefficient and coefficient of determination for different parameters in $FeCl_3$ at different pH.

Parameters	**Removal (%)**		**Correlation coefficient (ρ)**	**Coefficient of determination (R^2)**
	Min	**Max**		
BOD	8	12	0.943	0.8643 (Linear)
COD	15	61	0.832	0.7888 (Linear)
TS	-	-	-0.740	0.5474 (Linear)
pH	3	53	0.953	0.9088 (Linear)
TDS	2	39	0.973	0.9466 (Linear)
Salinity	0	22	0.974	0.9486 (Linear) Appendix 3.39
Conductivity	5	22	0.970	0.9413 (Linear) Appendix 3.39
Total P	18	77	0.908	0.8252 (Linear)
Hardness	7	32	0.990	0.9805 (Linear)
Alkalinity	10	37	0.971	0.9424 (Linear)
Sulfite	10	30	0.943	0.8892 (Linear)
Colour	3	99	0.819	0.6711 (Linear)
Turbidity	31	100	0.761	0.7444 (Power)
TSS	39	100	0.749	0.9768 (Power)
Ammonical-N	0	27	Insignificant	-
Cu	6	52	0.923	0.8522 (Linear) Appendix 3.40
Zn	39	64	0.859	0.8459 (Linear) Appendix 3.40
Ni	2	47	0.901	0.9566 (Power) Appendix 3.41
Co	1	61	0.837	0.8006 (Linear) Appendix 3.41
Mn	6	47	0.990	0.9806 (Linear)
Cr	18	48	0.839	0.7799 (Power)
Ca	3	35	0.946	0.9629 (Exponential)
Mg	4	12	0.867	0.7512 (Linear)
Cd	23	75	0.989	0.9781 (Linear)
Ba	4	80	-0.979	0.9685 (Logarithmic)
Pb	58	90	-0.961	0.9125 (Linear)
Al	6	56	0.999	0.9983 (Linear)
Sr	3	49	0.836	0.9228 (Power)
Li	1.4	5	Insignificant	-

Total solid was found to increase as the system become more acidic. This resulted from the generation of more flocs as the pH reduced. In acidic environment, the bonding of various elements within the system become more effective due to the competition among OH^- and other ions resulting with the production of more solid components (Zhang and Wang, 2009).

3.3.6.2 Physico-chemical treatment using alum.

The performance of alum in the removal of pollutants is less effective than that of $FeCl_3$ treatment. However, the total solid generation was found to have significant linear correlation with the concentration of alum (ρ=-0.805) as indicated in Figure 3.28.

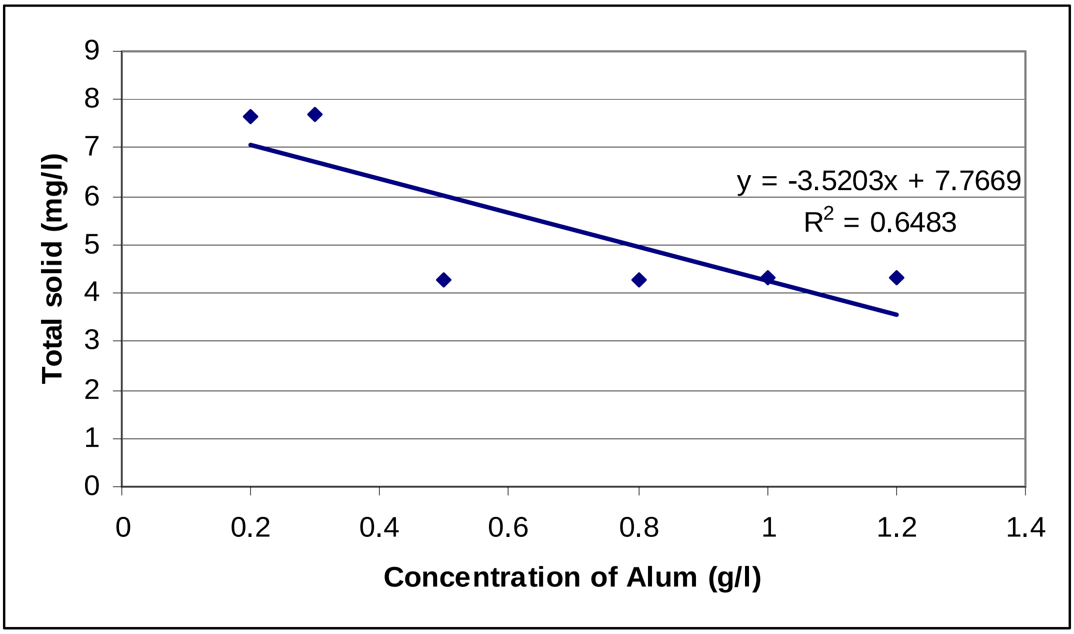


Figure 3.28: The total solid of leachate treatment at different concentration of alum.

Less solid was formed with the increase in alum concentration. No other significant trends were observed in terms of pollutant removal efficiency with the concentration of alum. Conductivity and salinity were found to have significant correlation with TDS of the leachate system where both display a linear reduction with the increase in TDS (Figure 3.29). This probably is due to the presence of more dissolved metal salt that the conductivity and salinity increased accordingly. Removal of Total Suspended Solid (TSS) ranged from 59 to 69%. However, no significant trend was observed. Turbidity was found to have significant linear correlation (ρ=-0.749) with TSS (Appendix 3.42) and final pH (Figure 3.30). It indicates that final pH increases with the increase in TSS. Less acidic environment must have cause the generation of more suspended solid in the system. On the other hand, increase in TSS resulted with higher turbidity of the system.

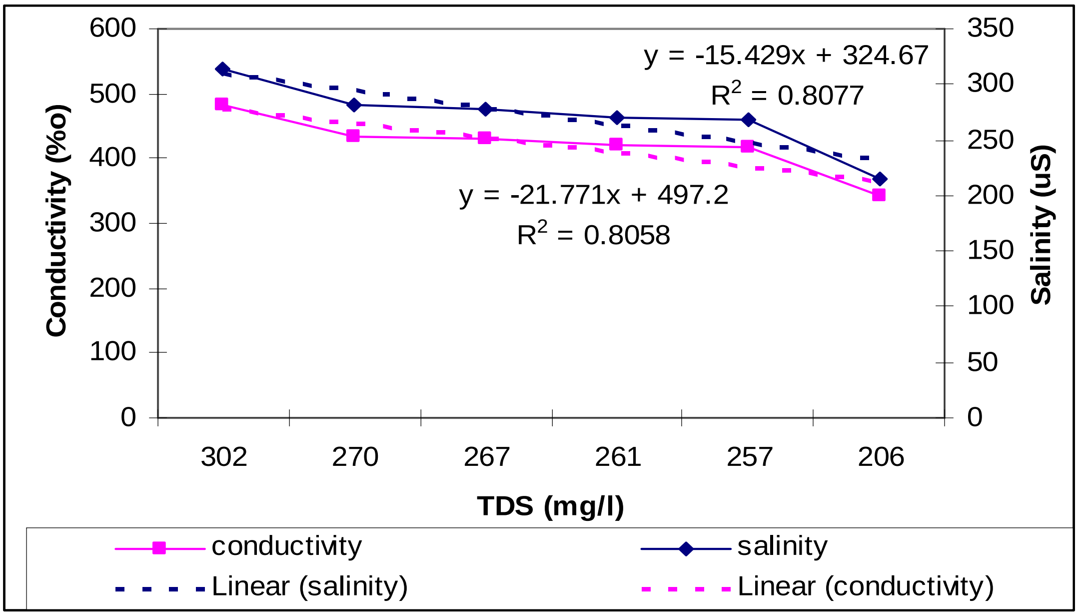


Figure 3.29: Linear trends in conductivity and salinity with the increase in TDS in leachate treated with alum.

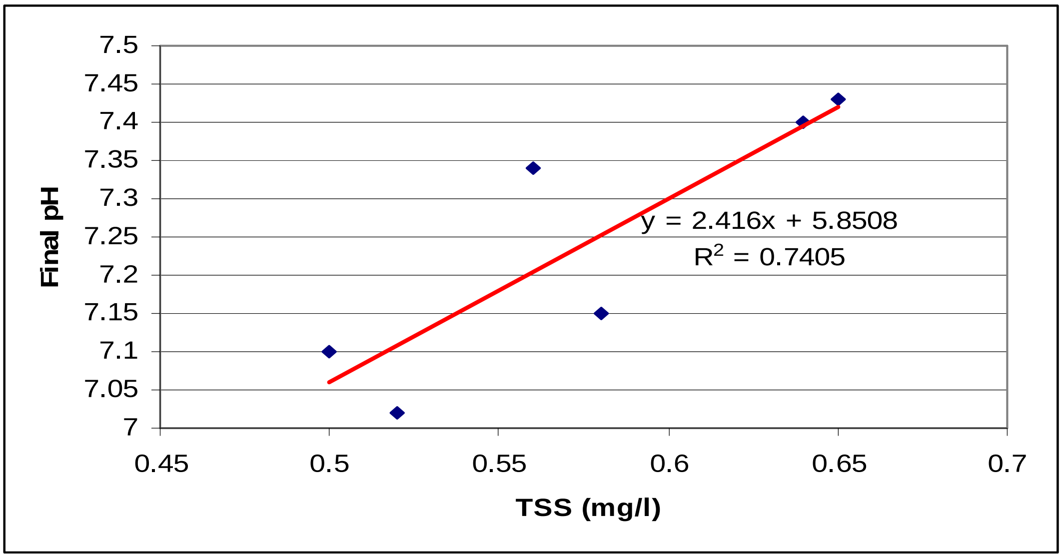


Figure 3.30: Linear trends between pH and TSS in leachate treated with alum.

The value of pH was found to have significant effects to the removal efficiency of various parameters. The following sections discuss the results and observation obtained from the study of leachate treatment by alum at different pH. The pH value of the leachate did not change much even after the coagulation-flocculation process completed (Appendix 3.43). Removal efficiency of TSS and TDS ranged from 14% to 92% with the increase in pH value (Figure 3.31 and Figure 3.32). More TSS and TDS are removed when the leachate become more alkaline.

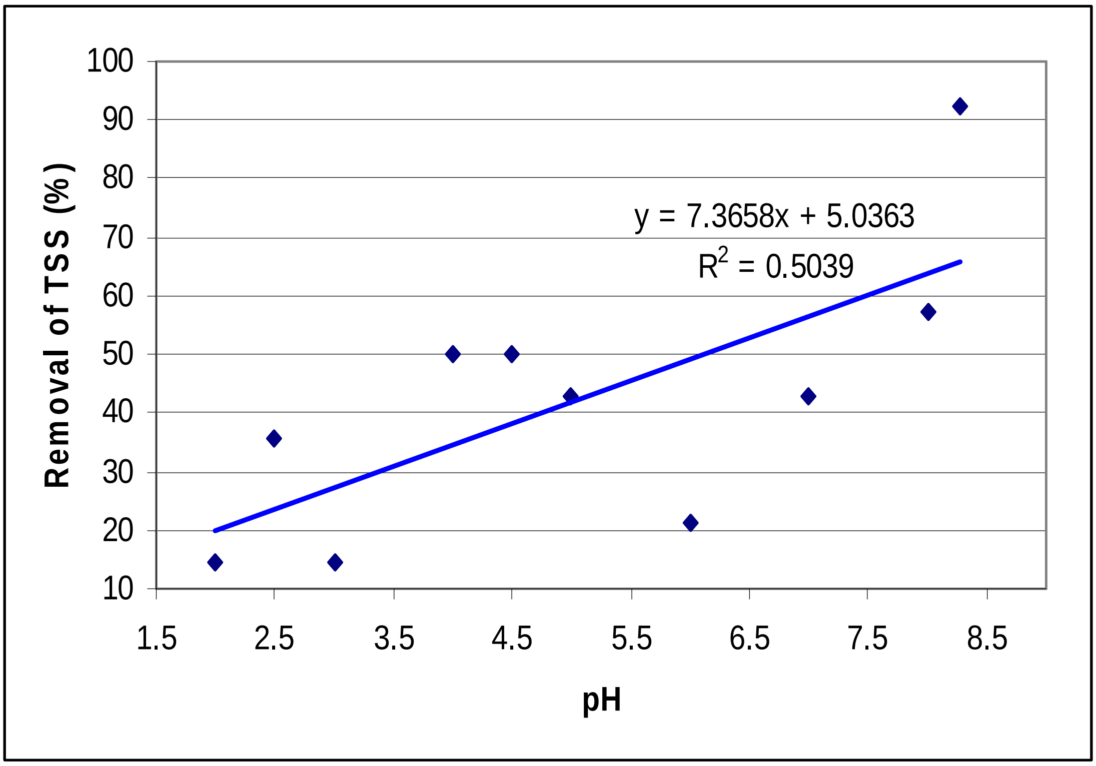


Figure 3.31: Removal efficiency of TSS in leachate treated with alum at different pH.

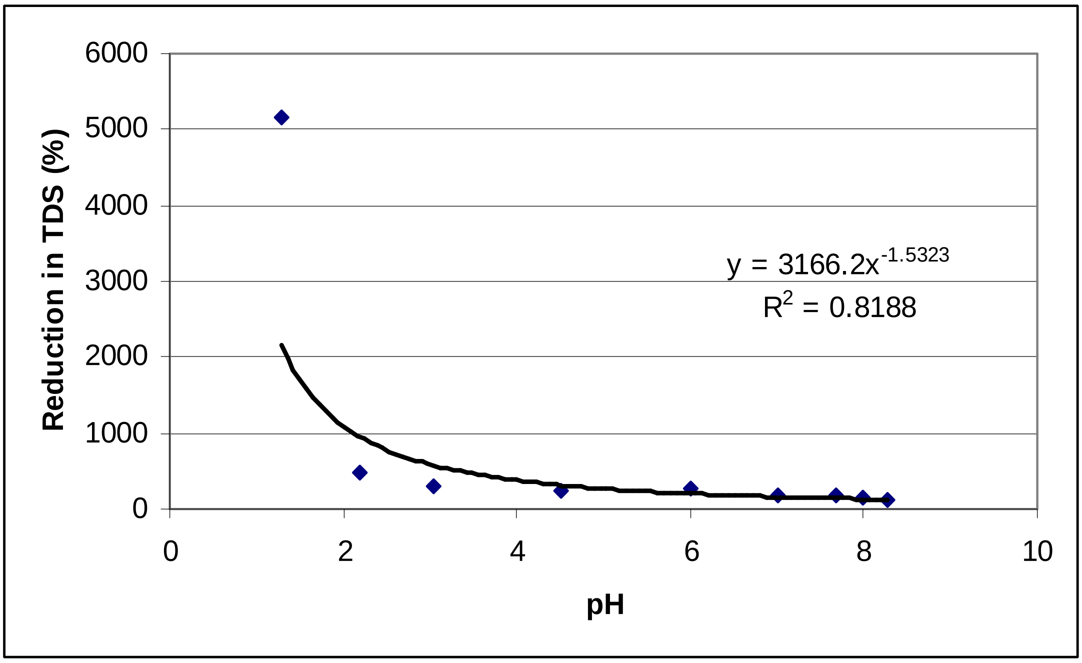


Figure 3.32: TDS in leachate treated with alum at different pH.

Removal of TDS was found to have a correlation equation: $y = 3166.2x^{-1..5323}$. TDS was found to have a linear correlation with conductivity as observed in leachate treated with different alum concentrations. No other significant correlations were derived from the analysis of the leachate treated with alum at different pH. Therefore, comparing alum and $FeCl_3$ to treat Malaysian landfill leachate, the utilization of $FeCl_3$ can produce more satisfactory results at suitable concentration and pH level.

3.3.6.3 Biological treatment

Microbial media exposed to leachate indicated very high colonies forming unit (CFU) on NA and PDA plates. The bacterial colonies were purified and inoculated into three Nutrient Broths (NB), namely NB1, NB2 and NB3, while fungal colonies were inoculated into Potato Dextrose Broth (PDB) i.e. PDB1, PDB2 and PDB3. Inoculation was random since microbial identification is not carried out. Results obtained from the biological treatment of the landfill leachate are discussed in the following sections. The addition of the bacterial cocktail managed to reduce 35-41% BOD, 69-73% COD and 30-45% total P, shown in Figure 3.33.

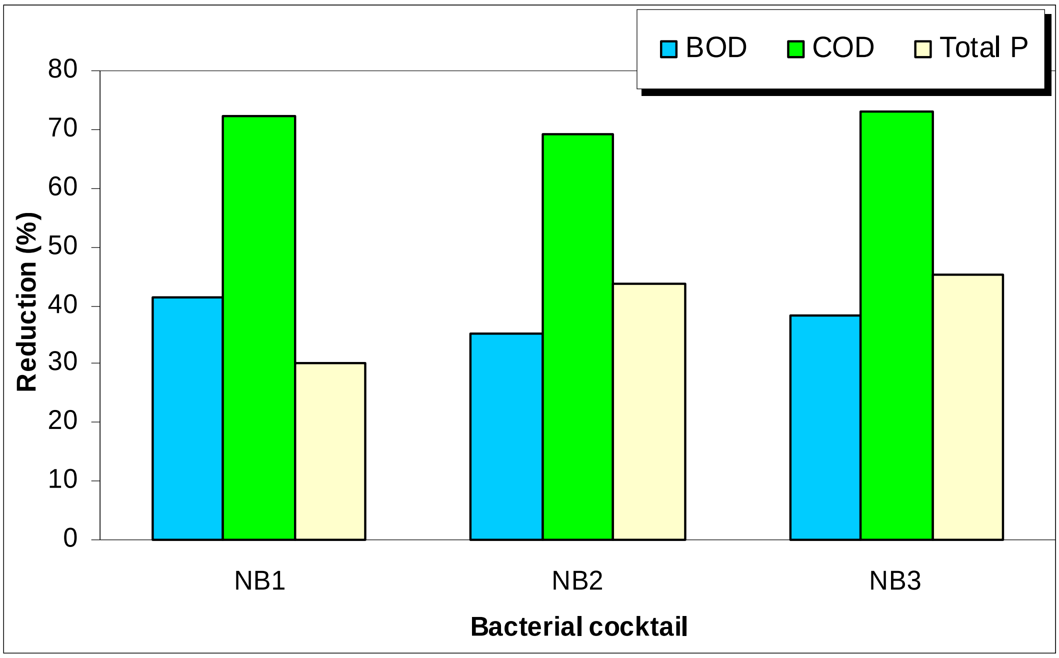


Figure 3.33: Reduction of BOD, COD and total P by different bacterial cocktails.

This is due to ability of the microbes to utilize the nutrient present for their metabolism. Ding *et al*. (2001) also had reported the ability of various types of microbes isolated from extreme environment to reduce various pollutants in the leachate. No significant difference is observed among the three types of bacterial cocktail. On the contrary, the removal of TDS, turbidity, salinity and ammoniacal-N were the highest was by NB3 at 54%, 54%, 23% and 12%, respectively. Figure 3.34 indicates the removal efficiencies of the three bacterial cocktails where NB3 is the highest. This probably is due to the presence of certain microbial strain in NB3 with higher ability to reduce these pollutants.

Many researchers including Zhang *et al*. (2009), Wichitsathian, (2004) and Isaka *et al*. (2007), had reported the ability of different microbes to degrade various pollutants such as colour and ammoniacal- N.

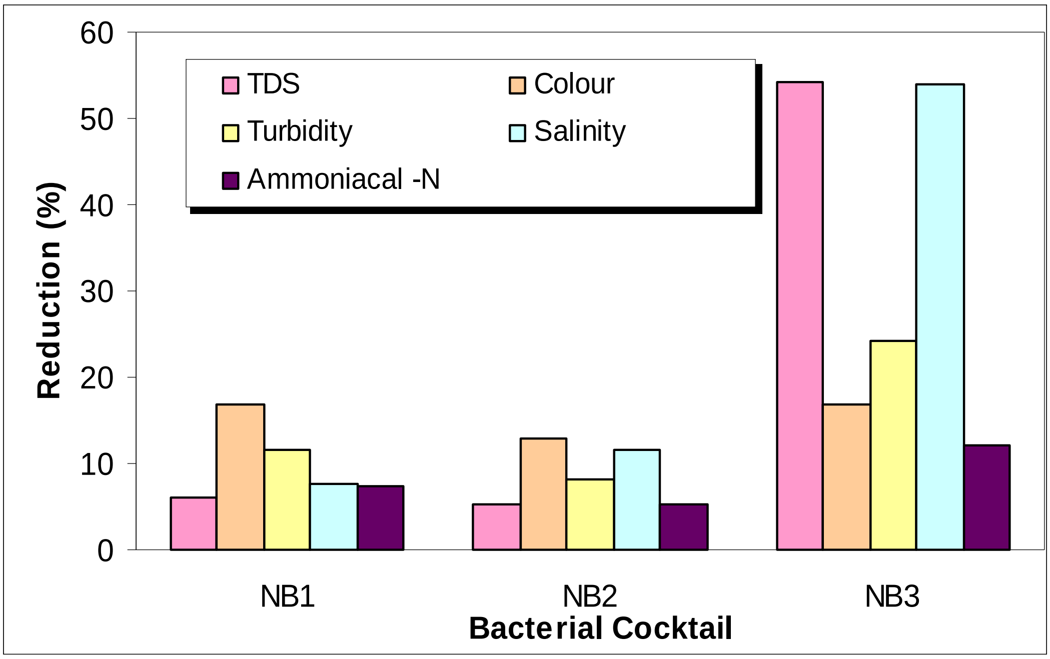


Figure 3.34: Removal efficiency of various pollutants by different microbial cocktail.

Figure 3.35 depicts the highest removal efficiency of Cd, Ca, and Co was by NB3 while for Pb the highest removal was by NB1.

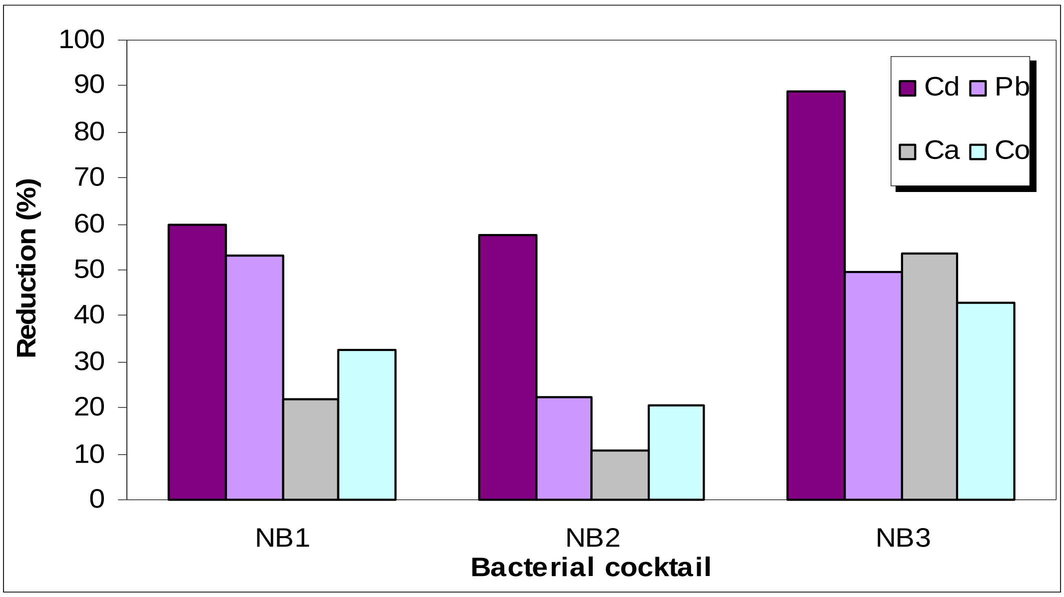


Figure 3.35: Removal efficiency of Cd, Pb, Ca and Co by different microbial cocktail.

Similarly, the removal of Mg, Ba, Mn, Ni, Cu, and Cr, was the highest for NB3 followed by NB1 and NB2 (Figure 3.36 and Figure 3.37).

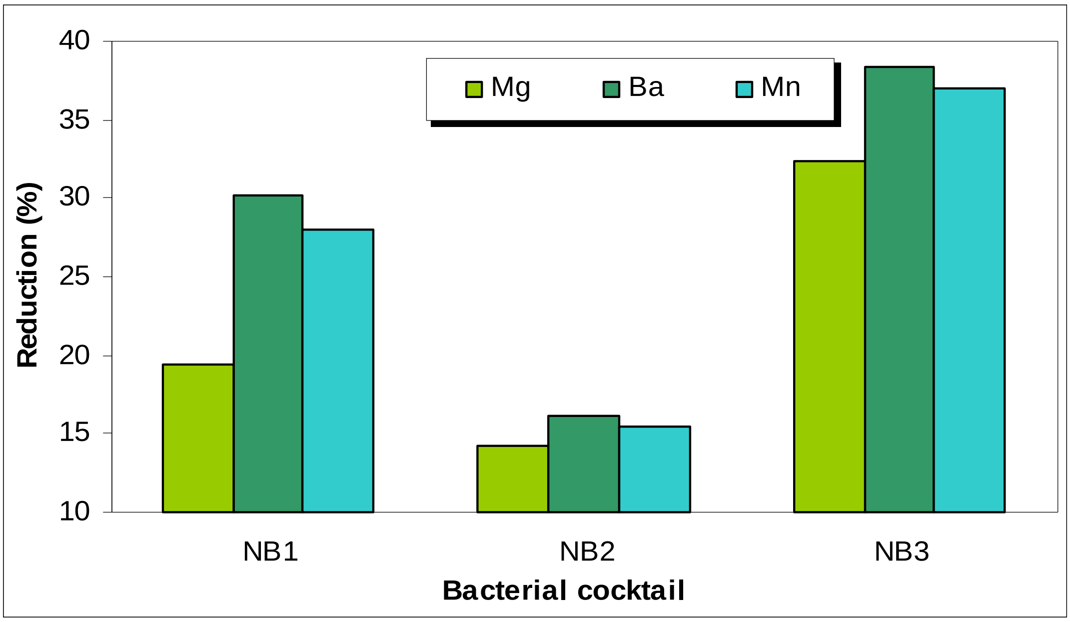


Figure 3.36: Removal of Mg, Ba and Mn from leachate treated with different bacterial cocktail

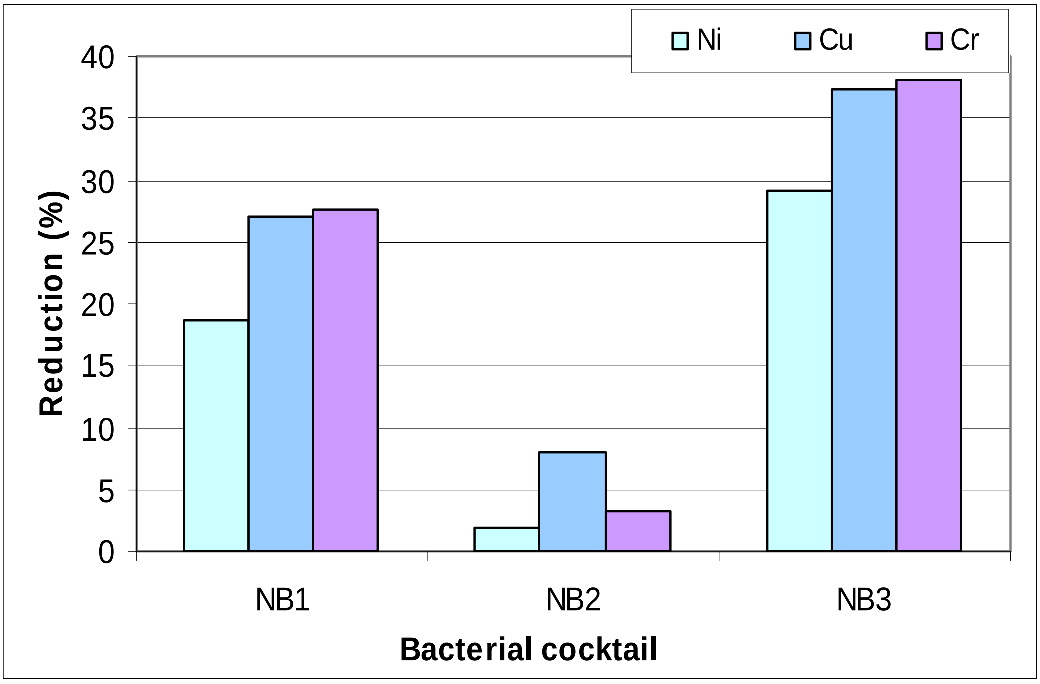


Figure 3.37: Removal of Ni, Cu and Cr from leachate treated with different bacterial cocktail

The ability to remove metal elements varies between the bacterial cocktail because different bacteria has different nutrient requirements for their metabolism. This resulted with different rate of metal elements uptake by the bacteria from their environment (Sierra-Alvarez, 2009; Pan *et al*., 2007: Ziagova *et al*, 2007). Consecutive sections discuss the efficiency of fungal cocktail in removing pollutants from the leachate.

Figure 3.38 illustrates the reduction percentage of TDS, salinity and total P by different fungal cocktail. PDB2 was found to have the highest removal TDS and salinity efficiency as compared to the others while PDB2 has the highest total P removal efficiency.

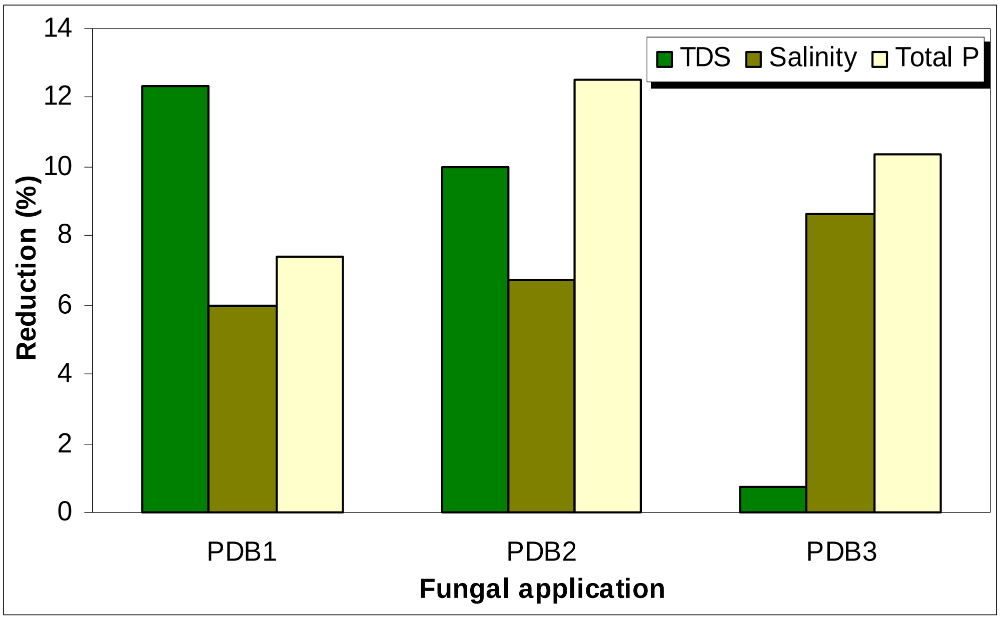


Figure 3.38: Removal efficiency of TDS, salinity and total P by different fungal cocktail.

On the contrary, no significant different was observed in the COD, BOD, conductivity, turbidity and ammoniacal-N removal efficiency. Figure 3.39 illustrates the efficiency of different fungal cocktail in removing selective metal elements.

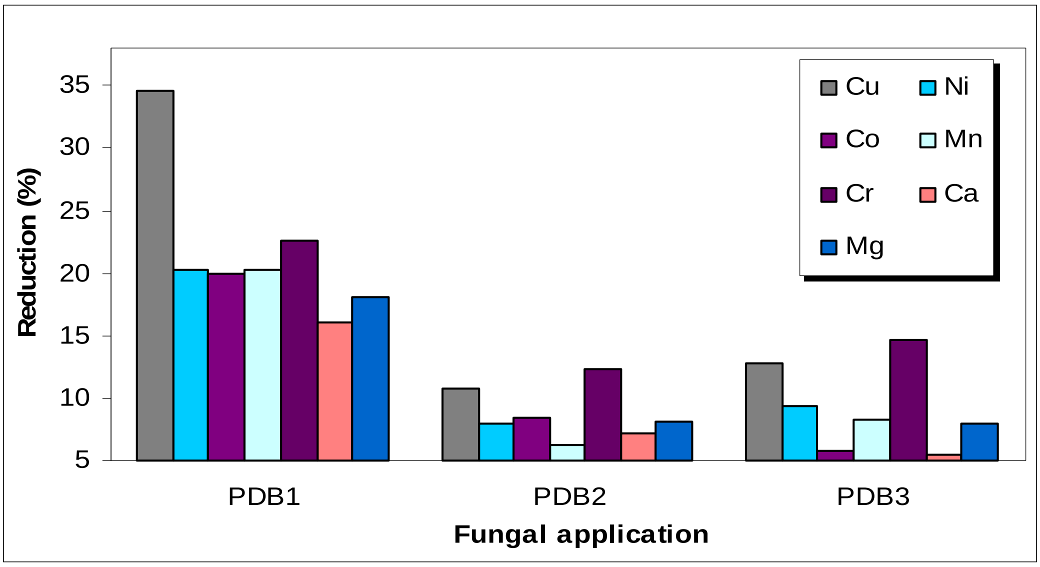


Figure 3.39: Removal efficiency of selected metal elements by different fungal cocktail.

It can be concluded that the bacterial cocktail works more efficiently in removing most pollutants than that of fungal cocktail. The integration of biological system into a physico-chemical and would enable higher better leachate treatment with also at a slightly reduced cost due to the optimal amount of chemical used (Mehmood *et al*., 2009).

3.3.7 Methane Oxidation Studies

Consecutive paragraphs discuss the results from the methane oxidation studies using landfill soil with changes in temperature and moisture content.

3.3.7.1 Methane oxidation by different soil types

Soil materials collected from the landfill displayed different trend in methane oxidation. Soil from active cell oxidized CH_4 the fastest followed by the 8-months and 1-year cells. For soil from the active cell, complete CH_4 conversion occurred within nine days with the speed accelerating within 47 to 218 hours. On the other hand, soil from 8-months and 1-year cells required 21 to 26 days to complete the process. The rate of CH_4 oxidation is due to the presence of microbial community within the biologically active cover i.e. soil system (Stern *et al*., 2007). The presence of organic compound in the soil would affect the microbial communities which determine the rate of CH_4 oxidation (Insam and Wett, 2008; He *et al*., 2008; Ritzkowski *et al*., 2006; Prant *et al*., 2006). The soil from the active cell generally has higher nutrient supply as compared to that of the 8-months and 1-year cells. Figure 3.40 depicts percentage of CH_4 over time by various types of soil.

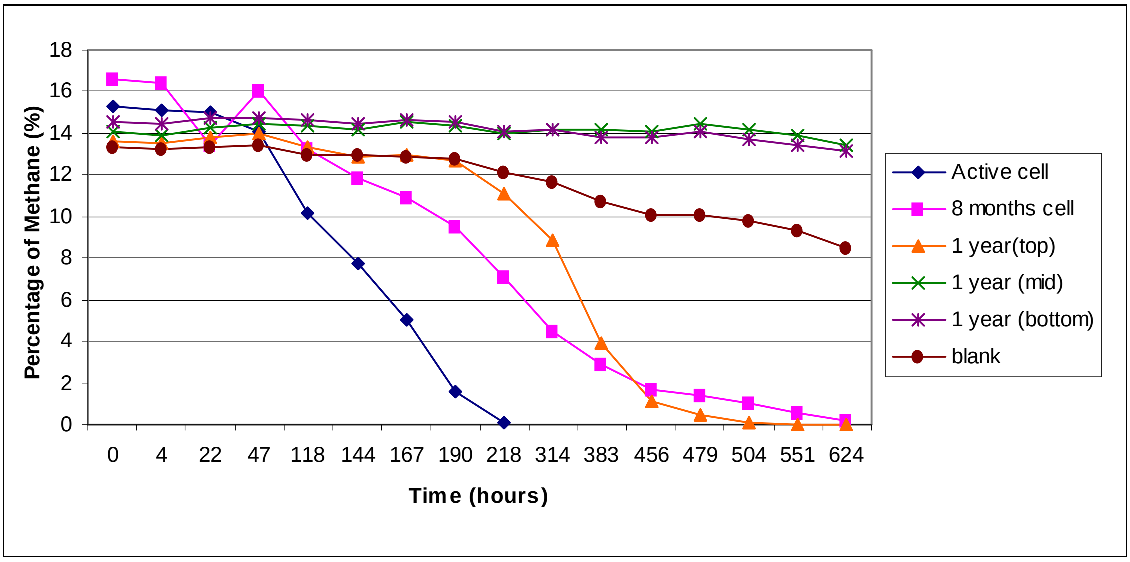


Figure 3.40: Percentage of methane over time by different types of soil.

Figure 3.40 indicated that depth of the soil has very minor influence in the ability to convert CH_4. Surface soil materials have higher potential in oxidizing CH_4 that three types of surface soil displayed better oxidizing rate (Table 3.25). This probably is contributed by the presence of abundant oxygen on the surface layer that the methanotrophs adapt better as compared to the deeper layers (Rich *et al*., 2008; Jugnia *et al*., 2008; Prantl *et al*., 2006). In addition, the analysis of oxygen concentration also indicated that the utilization of the top soil is better than deeper soil (Appendix 3.44).

Also observed in the study was the various rate of CO_2 generation among the different soil types. Soil from the active cell generated CO_2 at the fastest rate due to its capacity to convert CH_4 (Appendix 3.45). Active cell has been reported to have the highest capacity in generating CO_2 from CH_4 due to its chemical and biological characteristics (Sormunen *et al*., 2008; Spokas *et al*., 2005). Based on the three gas analysis, soil from the active cell was found to have the highest capacity in converting CH_4 at the fastest speed. Table 3.25 details the rate of CH_4 oxidation, O_2 utilization and CO_2 generation by the various soil types.

Table 3.25: Rates of process for CH_4, O_2 and CO_2

Soil Materials	CH_4 oxidation (% per hour)	O_2 utilization (% per hour)	CO_2 generation (% per hour)
Active cell	2.1323	2.6154	2.7205
8-months cell	1.2746	1.4225	0.7452
1-year cell (surface)	1.1683	2.0029	1.191
1-year cell (20 cm deep)	0.0196	0.2482	0.1293
1-year cell (30 cm deep)	0.0917	0.1443	0.0635
Blank	0.3366	0.2111	0

Based on the findings soil from the active cell was further tested by adjusting the temperature and moisture level.

3.3.6.2 Methane oxidation at different temperature

Since CH_4 oxidation involves microbial conversion of CH_4 to CO_2, temperature plays a major role in process. The results of the temperature study indicated that the rate of CH_4 oxidation increases with the increase in temperature. Figure 3.41 indicates at lower temperature (4°C to 15°C) CH_4 oxidation was very low.

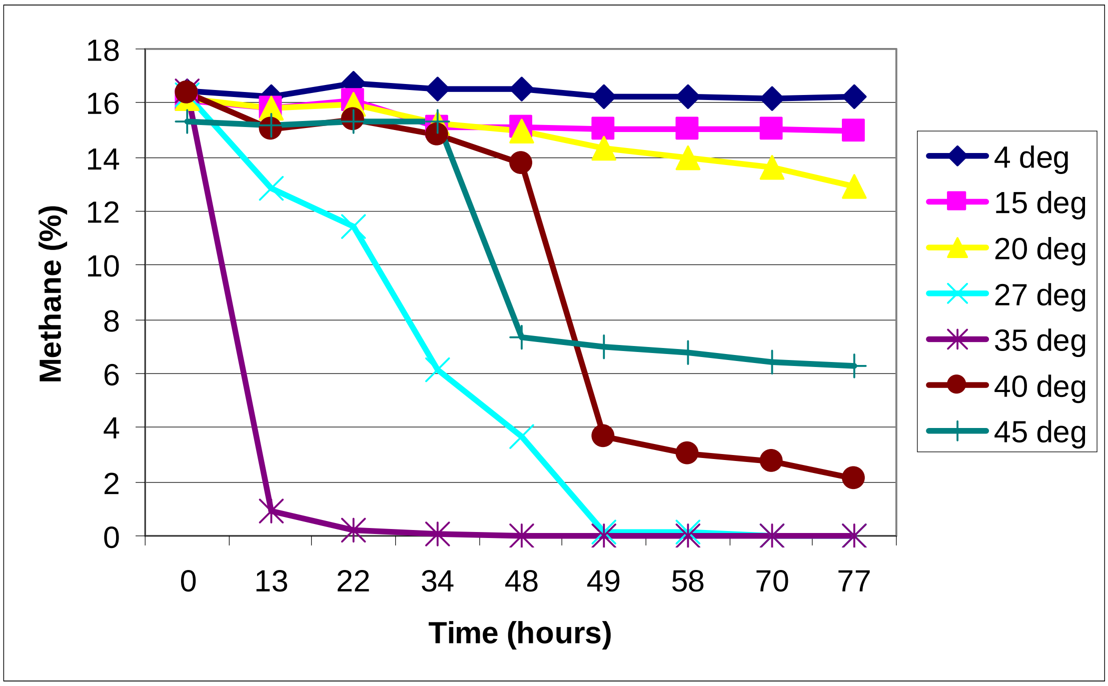


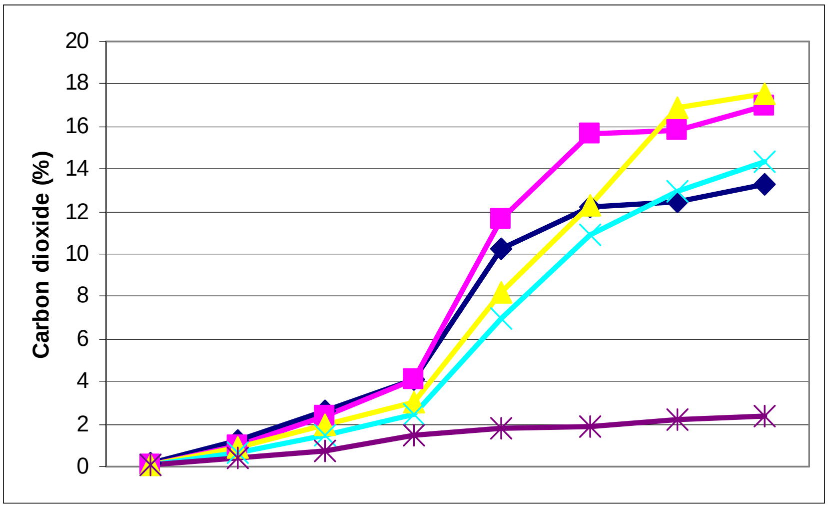
Figure 3.41: Percentage of methane over time at different temperature.

The low CH_4 oxidation level is generally due to the lack of microbial activities. This is so as the activities of cell tissues decrease when the temperature dropped due to the inactivation of enzymes in the cell tissues. This inhibits the activity of most microorganism particularly the methanotrophs (Einola *et al.*, 2008; Jugnia *et al.*, 2008; Alla *et al.*, 1993). However, at higher temperatures of 27 °C and 35 °C, the process occurred at rapid rate allowing complete CH_4 oxidation within 70 hours and 24 hours, respectively. Increase in temperature to beyond 35 °C indicated slower rate of CH_4 oxidation. This phenomenon could be contributed by the fact that the enzymes within the microbial system probably has undergone protein denaturation with only some enzyme that can tolerate higher temperature remain functioning (Han *et al.*, 2008; Iyer and Ananthanarayan, 2008).

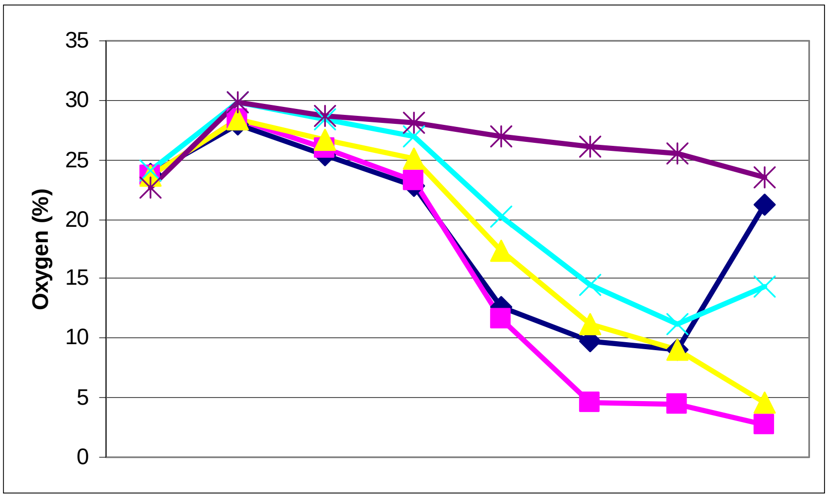
3.3.7.3 Methane oxidation at different moisture content

Results indicated that CH_4 oxidation occurred at different rate at different moisture level. This could be due to the fact that microbial community that converts CH_4 to CO_2 is highly dependent to the availability of moisture in the environment. The appropriate level of moisture content has the ability to enhance the rate of CH_4 oxidation. It also allows faster utilization of O_2 for metabolic activity which directly increases the CO_2 concentration. The rate of methane oxidation was best when moisture as much as 5% to 10% were introduced. The lower or higher than the ranges would result in the slowing of the oxidation process. Figure 3.42 (a), (b) and (c) illustrate the concentration of CH_4, O_2 and CO_2 at different moisture level.

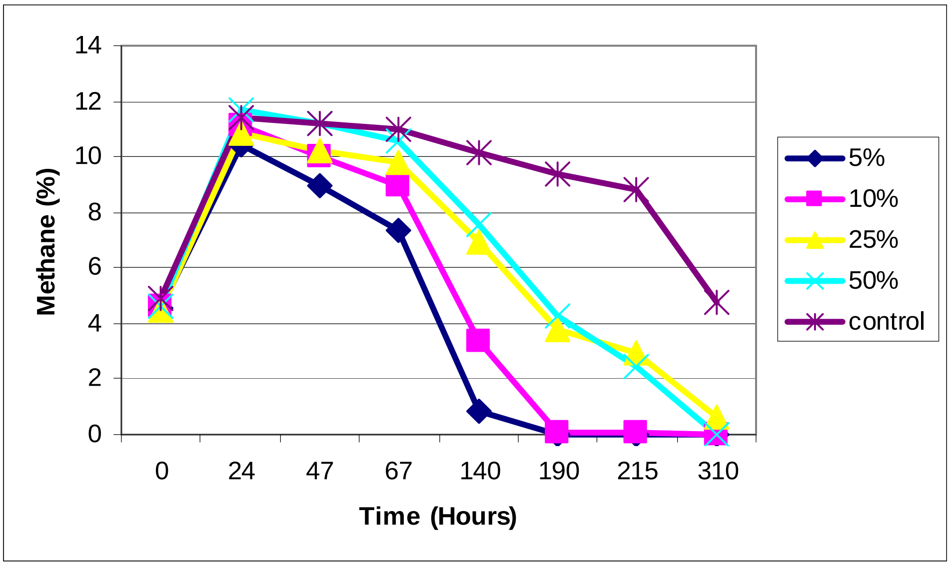
Methane oxidation came to a completion after 8 days for soil samples with 5% and 10% moisture content. At this range (5-10%), the availability of H_2O probably is at the best level to suit the needs of the methanotrophs for CH_4 oxidation (Chen *et al*., 2008). Too low moisture content inhibited the oxidation process to almost zero level while too high moisture content caused the environment to be converted to anaerobic condition. Higher percentage of moisture hinders the process causing CH_4 oxidation process to slow down (Einola *et al*., 2008; Chen *et al*., 2008; Bogner and Spokas, 1993).



(a) CO_2



(b) O_2



(c) CH_4

Figure 3.42: Concentration of gases over time at different moisture content.

Based on the studies, the most suitable material to promote methane oxidation is soil from active waste cell. Suitable temperature (35^{0}C) and moisture level (5%) would expedite the process to reduce methane emission from landfill.

3.4 Conclusions

Results indicated that the largest waste component was organic waste by which diverting it via vermicomposting would reduce more than half of the total waste sent for landfill disposal. On the contrary, the indiscriminate disposal of the MSW into landfill will result with the generation of leachate and landfill gases. The characteristics of leachate from the study differed from one landfill to the others. On average, the pollution intensity was below the Standard B with the exception to Cr concentration. However, with the application of physico-chemical treatment pollutants in the leachate were reduced significantly. Biological treatment reduced the leachate pollution intensity further that all parameters were below the limit stipulated in Standard B (EQA 1974). On the other hand, landfill gas mitigation was conducted via methane oxidation trials. The most suitable media for methane oxidation was the soil from active waste cell at 30-40^{o}C and 5-10% moisture content. The study indicated that current landfill management system could be improved to minimize the environmental burden from waste disposal.