Effect of Vegetation in a Dissolved Metal Passive Compost Stormwater Treatment System

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by

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Abstract

A better understanding of the flow and sorption properties of vegetated compost roadsides is needed before such passive, low cost treatment beds can be recommended as a Best Management Practice (BMP) to treat metals in stormwater runoff during the non-vegetative growth and high rainfall period between Oct. and April in the Pacific NW. The effect of four different flow rates between 70 and 900 mL/min on retention time in 12° sloped compost and grassed compost beds (1.02 m x 0.254 m x 0.10 m deep) were examined using bromide as a conservative tracer. Zinc sorption (at 3.0 mg/L, pH=5.6, 11°C) buffering potential and turbidity levels were examined in both beds at 264 mL/min. The mean particle size of the compost was 0.0063 m.

Roots that had amassed for 7 months prior to the experiments increased retention time only at the low flow rate of 70 mL/min. At this flow rate, most of the flow was concentrated along the bed bottom where the root mass was thickest. Excepting at the two lowest flow rates, the drainable water volume in the compost bed was higher than in the grassed compost bed. Bed composition in duplicated trials did not affect zinc breakout at 264 mL/min when retention time and percentage of the bed utilized were similar in both bed types. Breakout time in the beds was approximately 230 hr. Both bed types buffered influent pH to as high as 7, and this ability decreased over time (F=50-200, p<<<0.0005). Initial turbidity readings for both treatment beds were found to be significantly higher than the runtime turbidity. The grassed compost bed produced lower initial and runtime effluent turbidities (tstat=3.18-. 5.14, p=.0005 to <<0.0001). Results indicated that grassed compost beds are as effective as compost beds at a flow rate of 264 mL/min for water holdback, pH buffering and zinc metal sorption, but grassed compost bed effluent would likely be clearer. Results will be used to design BMPs for industrial sites that have similar flow and zinc runoff levels. Future research should focus on different metal concentrations and species.

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I dedicate my thesis to DP. Without your support I would have been doomed to an existence of RezLife and ramen noodles… I am forever indebted to you.

1.0 Introduction

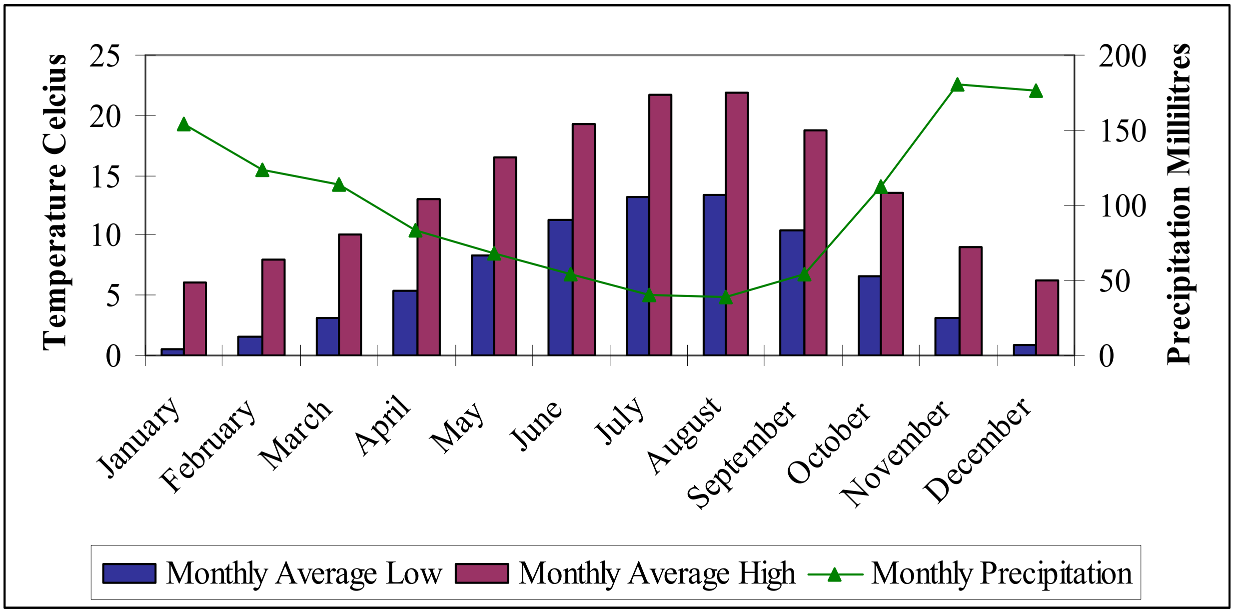
Highway stormwater runoff is a persistent and often overlooked problem worldwide Stormwater delivers a wide variety of environmental contaminants at varying concentrations in sporadic, non-point source flow to isolated often inaccessible stretches of roadway. Pollutants on highways can be attributed to both anthropogenic and environmental factors including vehicle wear and emissions, geographical locale, season, highway surface composition, weather, local land use conditions, average daily traffic and highway maintenance practices (McKenzie and Irwin 1983; Kobriger and Geinepolos 1984). Metals such as zinc, lead, copper, chromium, cadmium and iron are common in highway runoff, and they are typically derived from automotive and infrastructure abraded material (Noll and Miller 1975). Without treatment methods in place, contaminants enter waterways, depending on concentration, can cause fish kills, degrade drinking water, and diminish water-based recreation and tourism opportunities. As well, economic losses to commercial fishing and aquacultural industries, lowered real estate values, and damage to habitat of fish and other aquatic organisms have been attributed to contaminants in runoff. Further, the inevitable costs of clean-up and pollution reduction and reduced aesthetic values of lakes, streams and coastal areas all show the environmental, social and economical impacts of runoff (British Columbia Ministry of Water, Land and Air Protection 1995).

Dissolved heavy metals in urban runoff are of particular concern because they enter the aquatic environment and accumulate in living tissues, working their way up the food chain and eventually affecting human health (ERMD 2005; Volesky 2001). There is also an increase in the chronic long term contamination of sediments and the food chain through the release of persistent, bio-accumulative toxic agents (US EPA 2002). Locally, twenty out of forty salmon-bearing streams in the Fraser-Delta Habitat region near Vancouver, B.C. are degraded due to urban storm water runoff (Nener and Wernick 1997). An even larger example of the problem is that 40% of the lakes, rivers and coastal waters monitored by the United States do no meet water quality goals (US EPA 2002).

Current methods of stormwater metal treatment are either considered too capital intensive, too costly to maintain, to require too much land and/or to be ineffective at removing metals below environmental guidelines. Another problem with current methods is that they as often require runoff to be redirected to a single point of treatment, tend to destroy the look of the natural environment and/or involve hidden costs (Minton 2002; Volesky 2003). The roadside applications of compost and vegetative filter strips are emerging as a viable alternative to traditional cost intensive methods of stormwater treatment. In this application, the runoff from the roads passively via gravity moves across the road surface to be intercepted by a ditch. The ditch faces intercepting the water is lined with compost and/or vegetated compost. Hence, the application relates to ditched roadways or impervious areas surrounded by a ditch. The practice of adding compost to vegetative strips is becoming increasingly more common in order to decrease erosion (by holding back water to reduce peak flows, bindings soil particles, etc) and increase roadside vegetation establishment as well as its use in stormwater treatment (Larimore and Balzer 2007). To date, the effectiveness of composted and vegetative roadsides filter strips has been investigated under a limited set of conditions for treating roadway runoff, and no research has been reported on the effectiveness of the use of the combination of vegetation in compost roadside strips for treating roadway runoff. Many best management practices (BMP) for controlling stormwater runoff using vegetation for treatment and flow control were based on model predictions developed for warm-weather climates subject to summertime rainfall events. However, these data have been considered too limited to be useful for predicting performance over a wide range of environmental and roadway conditions. (Davis et al. 2009) has identified the need to design vegetation selection, media composition and configuration that are appropriate to the range and limitations of local conditions. Additionally, information on the horizontal flow of runoff through the filter strips at different flow rates to determine retention time of filter is required. A better understanding of the metal sorption capacity, subsurface hydrodynamic conditions and longevity in vegetated and/or compost roadside strips or filters is also needed before they can be recommended as a BMO to treat metal in highway runoff.

This research focuses on the performance of roadside compost filters and vegetative strips function in the temperate rain forest region that extends from central coastal California to South Alaska. In this region, moisture laden winds move inland from the Pacific starting in the fall, meet the barrier of the coast range, and then rise abruptly. Suddenly cooled by this upward thrust into the atmosphere, the moisture in the air is released as rain and snow in amounts up to 635 cm by the end of March. During the summer, when the winds shift to the northwest, the air is cooled over chilly northern seas and rainfall is lower. Average monthly temperature and rainfall vary within the long temperate rainforest region. In the Greater Vancouver Regional District (GVRD), British Columbia, the rainy season tends to begin in October and ends in April (see Table 1). The average monthly rainfall and temperatures can also be found in Table 1.

Table 1- Climate Data for GVRD* (Source-Environment Canada)



***Climatological information is based on monthly averages for the 30-year period 1971 - 2000**

***Mean number of precipitation days = Mean number of days with at least 0.2 mm of precipitation**

***Precipitation includes both rain and snow**

The low winter temperature tends to stop grass from growing along roadsides in much of the temperature rainforest. Under these conditions, the biological activity of vegetation receiving the runoff during this period is significant below summer expectation levels. Hence, the cooler heavy rainy season weather pattern in much of the temperature rain forest present additional challenges to the selection, design, and maintenance of stormwater treatment BMPs.

Thesis Layout:

An overview of stormwater as an environmental threat with emphasis on current treatment technologies is presented in Chapter 2. Yard-waste compost and its potential for removing dissolved metals from stormwater is introduced in more detail, followed by a description of sorption by humic compounds. The effectiveness of vegetative strips on roadway runoff is also discussed. Chapter 2 finishes with a review of chemical breakout-time and residence-time distribution theory.

The specific scope and objectives of the thesis project are presented in Chapter 3. Chapter 4 details the materials and methods used in the research starting with the selection vegetation and sorbent media. Chapter 4 then is used to describe the experimental equipment design and the laboratory influent compositions including the rational for the choice of metal and its concentration. This is followed by details of the bromide tracer experiments, zinc sorption capacity experiments and pore volume experiments and includes descriptions of the analytical and measurement techniques used throughout the research project.

Chapter 5 presents the retention time results of the bromide tracer experiments at different flow rates. In addition, the zinc concentration, pH and turbidity exiting from experimental sloped compost and compost/vegetated beds that had received a steady supply of a low level zinc solution are presented. Bed longevities and sorption patterns are also compared. T-test and slope comparison statistical tests are used to determine if the bed configurations are significantly different.

Chapter 6 is used to discuss the results of this work relative to existing literature and current knowledge related to these filters in stormwater management applications. Lastly, Chapter 7 is used to present conclusions and recommendations for future research.

2.0 Literature Survey

This chapter provides an overview on background information on stormwater and the role its constituents play as an environmental contaminant as well as current stormwater treatment alternatives. Recent research on the use of compost and vegetation as used in the roadside treatment of stormwater is then detailed. Finally, literature pertaining to sorption breakout curves and residence time distribution is reviewed.

2.1 Stormwater as Non-Point Source Pollution

Highway runoff is a typical non-point source (NPS) pollution and is difficult to monitor and control as a result of its dispersed and variable nature. Non-point source (NPS) pollution has been identified as one of the major remaining sources of water quality degradation (Lee et al. 2002; British Columbia's Ministry of Environment, Land and Parks (Now Ministry of Water, Land and Air Protection) 2005). Highway runoff has been considered the highest source of NPS pollution (Kobriger and Geinepolos 1984).

The effect of urban runoff is becoming even more pronounced with the addition of more roadways, parking lots and other impervious surfaces (Brabec et al. 2002). Where natural vegetation and soil structure once allowed the gradual adsorption and through-put of rain and snowmelt, paved streets speed the delivery of water and pollutants such as metals, sediments, nutrients, toxins, pathogens, and oil and grease that have accumulated on the surface of the road during dry periods (ERMD 2005). The current trend shows that the presences of metal in the environment are more prominent today correlating to an increase in population rate (Volesky 2001). Daily, every mile of highway affects adjacent watersheds.

Impervious surfaces can accumulate nutrients (nitrogen and phosphorus) and metals during dry periods when little or no precipitation occurs (Sorme and Lagerkvist 2002). Upon the first substantial rainfall, a phenomenon known as the "first flush" occurs. The first flush is the initial period of storm water runoff (Usually lasting a duration of 30 min) during which the concentration of pollutants is substantially higher than those observed in the later stages of the storm (Lee et al. 2002). This high concentration of pollutants shocks the system which receives it and ultimately leads to the eutrophication of bodies of water and metal poisoning of the aquatic and marine ecosystems.

Pollutant loads on highways can be attributed to both anthropogenic and environmental factors and include but are not limited to vehicle wear and emissions, geographical locale, season, highway surface composition, weather, local land use conditions, average daily traffic and highway maintenance practices (McKenzie and Irwin 1983; Kobriger and Geinepolos 1984). Primary sources of metals are automotive and infrastructure abraded material (Noll and Miller 1975). Analyses of tire debris on urban paved surfaces revealed that tires can deposit 0.12 kg/km of highway surface for every 1,000 automobiles per day (Muschack 1990). More importantly, the amount of pollutants discharged with surface runoff is related more to the characteristics of the area, wind conditions, frequency of precipitation, the Antecedent Dry Period (ADP) (dry period between runoff events) and the volume of runoff and not directly dependant on the traffic frequency (Stotz 1987).

It is difficult to predict specific pollutant loadings in stormwater for a variety of reasons. Firstly, the concentration peak for different pollutants can vary during the same storm event and that the concentration load for the same pollutant can vary during different storm events in the same watershed (Gupta and Saul 1996). Additionally, during the first flush concentration spikes are highly variable (Deletic and Maksimovic 1998; Lee et al. 2002). The existence of a first flush in a catchment area must be considered when choosing the most appropriate treatment system.

Pollutants of concern on highways, their loadings and runoff concentrations are discussed below as well as common stormwater runoff rates.

2.1.1 Particulates

Solid pollutants found in stormwater can be categorized into particulates, metals, polychlorinated biphenyl (PCB), pesticides, pathogenic bacteria, de-icing agents, organic matter, nutrients and others such as asbestos, rubber and other special compounds (Gupta et al., 1981). Pavement wear, vehicles, maintenance of highways, atmospheric deposition and from surrounding land use activities contribute to particulates accumulating on highways and in the surrounding environment (Onwumere 2000).

The volume of sediment yield from a road depends on sediment supply and transport capacity (Anderson and Simons 1983). Sediment yield is determined by road geometry, slope, length, width, surface, and maintenance (Anderson and Simons 1983; Grayson et al. 1993), in addition to soil properties and vegetation cover (Horner and Mar 1983). Particulates facilitate the transport of metals and other contaminants which eventually incorporate into stormwater sediment. The percentage of metals associated to suspended solids is higher than the dissolved fraction (Barbosa and Hvitved-Jacobsen 2001). Suspended solids provide surfaces for metals to be adsorbed and thus serve as carriers for metal pollutants. Metals, polycyclic aromatic hydrocarbons (PAHs), phosphorus and organic compounds are adsorbed onto TSS (Rossi et al. 2005).

Typical suspended solids concentrations are summarized in Table 2 below:

Table 2- Concentration of Suspended Solids found in Runoff

Suspended Solids				
Range (mg/L)	Loading (kg/ha/yr)	Road Type	Location	Reference
122-369	3270	Highway	Burnaby, Canada	Onwumere 2000
38-154	681	Highway	Richmond, Canada	Onwumere 2000
137	-	Highway	Pleidelsheim, Germany	Stotz 1987
181	-	Highway	Obereisesheim, Germany	Stotz 1987
252	-	Highway	Ulm/West, Germany	Stotz 1987
180	-	Freeway	Milwaukee, USA	Shelly and Gaboury 1986
9-1,540	-	Residential	Fresno, USA	FMFCD 1984
8-4,300	-	Residential	Fresno, USA	FMFCD 1984
2-3,720	-	Commercial	Fresno, USA	FMFCD 1984
51-2,770	-	Industrial	Fresno, USA	FMFCD 1984
99	-	Freeway	U.S. National Average	Pitt 1985

2.1.2 Metals

Both dissolved and total metals are of particular interest in stormwater runoff due to their toxicity, ubiquity, and the fact that metals cannot be chemically transformed or destroyed. Highway runoff often contains the metals lead, zinc, iron, cadmium, chromium, nickel and copper at levels higher than background concentrations (Onwumere 2000). The metal concentrations found in stormwater runoff are considered low in comparison to concentrations found in industrial or mining wastewater which is can be found to be several hundred (Fe~200mg/L at a pH 2-3) to several thousand mg/L total metals (Kocasoy and Güvener 2008; Jørgensen 1979).

Typical metal concentrations found in runoff can be found in Table 3 below:

Table 3- Runoff Metal Concentrations (BC Approved Water Quality Guidelines, 2006)

Metal	Total Recoverable (mg/L)	Dissolved (mg/L)	BC Aquatic Guidelines* (mg/L)
Cadmium	<0.004-0.001	0.00068-0.002	1.3810E-05
Chromium	0.0085-0.015	<0.001	Currently under review
Copper	0.014-0.22	0.004-0.015	0.0030
Iron	1.6-9.05	0.089-0.37	0.3
Lead	0.0074-0.56	0.0018-0.025	0.0049
Zinc	0.01-0.91	0.043-0.19	0.033

* BC Aquatic Guidelines for the Protection of Aquatic Life based on a Hardness of 90 mg/L

Even the low concentration of metals in stormwater runoff have the potential to cause direct toxic damages to plants, animals (Pais and Jones 1997), and microorganisms (Aoyama and Nagumo 1997) growing in the receiving water bodies. There is a risk of the bioconcentration and biomagnification of metals in the aquatic community because metals do not break down or decay. Upon reaching the ecosystem, these elements undergo physical, chemical and biological changes such as adsorption onto clay particles, up-taken by plants or they remain in solution (Onwumere 2000). The biological activity of both toxic and essential metals depends on the ability of the ions to combine with other molecules and atoms and their speciation in solution (Onwumere 2000). Therefore, waters containing a high total metal concentration (including both dissolved and particulate metals) may be less toxic than waters containing a lower concentration of the metal in the ionic form (Onwumere 2000). For example, the toxicity of organically bound copper to aquatic organisms is much less than ionic copper (Ferguson 1990; Onwumere 2000).

Metals in the dissolved form exhibit different levels of mobility through both media and liquids. Ranking the order of mobility from most mobile to least mobile: zinc > lead > cadmium > manganese > copper > iron > chromium > nickel > aluminum (Burton and Pitt 2002). However, a partitioning analysis of dissolved and particulate bound metals from pavement sheet flow showed that zinc, nickel and copper are predominantly found in dissolved form while lead, iron, aluminum and chromium were predominately found particulate-bound (Mesquita and Carranca 2005; CDT 2005).

2.1.3 Organics

Organic contaminants commonly found in urban and highway runoff include oil and grease, aliphatic and aromatic hydrocarbons, plasticizers, polychlorinated biphenyls (PCBs), pesticides, polynuclear aromatic hydrocarbon (PAHs) (Onwumere 2000). Hydrocarbons found in road runoff may come from lubricating oils, fuel, exhaust emissions and road wear. The concentration of PAHs increases as more oil is used. PAHs dissolve from fuel directly into the oil or are the result of incomplete fuel combustion (Maltby et al. 1995). A summary of the commonly found organic pollutants is found in the tables below:

Table 4- Oil and Grease Runoff Concentrations

Oil and Grease Concentration (mg/L)	Source	Location	Reference
1.0-8.0	Residential	Fresno, USA	FMFCD 1984
0-5.0	Residential	Fresno, USA	FMFCD 1984
0-26	Commercial	Fresno, USA	FMFCD 1984
0-80	Industrial	Fresno, USA	FMFCD 1984

Table 5- PAHs Runoff Concentrations

PAHs Concentration (mg/L)	Road Type	Location	Reference
0.00261	Highway	Pleidelsheim, Germany	Stotz 1987
0.00297	Highway	Obereisesheim, Germany	Stotz 1987
0.00251	Highway	Ulm/West, Germany	Stotz 1987

In British Columbia, total PCB concentrations in freshwater sediments are generally below the detection limits (0.02 and 0.01 µg/g) (SEAM 1989; Swain and Walton 1988). However, elevated levels of PCBs have been measured in sediments off certain industrial facilities; e.g., up to 1.0 µg/g was measured in sediments adjacent to the Belkin Paperboard paper recycling plant in Burnaby (Garrett 1983). (Goyette and Boyd 1989) found that mean PCB concentration in sediments from Vancouver Harbour (based on 1985/86 data) ranged from < 0.02 µg/g to 0.90 µg/g. The maximum concentration

(0.90 μg/g) was recorded for the site at Burrard Yarrow on the north shore in the GVRD. Overall mean level for the Vancouver Inner Harbour (44 sites) was 0.17 +/-0.20 μg/g. Sediment PCB concentration in Port Moody Arm in 1985/86 (13 sites) averaged 0.06 μg/g (range, 0.02 - 0.18 μg/g), and in 1987 (33 sites) the average was 0.13 μg/g (range, 0.03 to 0.32 μg/g). For reference, the EPA's enforceable maximum contaminant level for PCBs in public drinking water systems is 0.0005 ppm. There is limited data on highway organic contaminants (Onwumere 2000).

2.1.4 Nutrients

Phosphorus and nitrogen are the nutrients of concern in stormwater. They are generally present in dissolved form and as a result, they cannot be settled out from runoff (Kobriger and Geinepolos 1984). The presence of nitrogen and phosphorus compounds in excessive amounts leads to excessive growth of aquatic plants, surface algal scum, water discoloration, turbidity, odor, and low concentrations or fluctuations of dissolved oxygen (Table 6 shows common chemical oxygen demand (COD) concentrations in stormwater runoff). The time between storm events can result in a build up of nutrients on surfaces, leading to high pollution loads. Generally, tree leaves fertilizers and lubricants are sources of phosphorus (Makepeace et al. 1995). Nitrogen occurs as organic nitrogen, NH^{4+}, NO^{3-} and NO^{2-}. NO^{3-} and NH^{4+} are the forms used by aquatic plants as nutrients. Organic nitrogen and nitrite are also included in pollutant accounting because these forms can be converted to the available forms. The dissolved forms of nitrogen can be toxic to aquatic organisms in excess. Nitrate is acutely toxic at concentrations as low as 5 mg/L to steelhead eggs, NO^{2-} at 0.19 mg/L to rainbow trout, and NH^{4+} at 0.0017 mg/L to pink salmon (Makepeace et al. 1995). Sources of nitrogen in stormwater include fertilizers, industrial cleaning operations, feed lots, animal excrement, and combustion of fuels (Makepeace et al. 1995).

Table 6- Chemical Oxygen Demand (COD) Runoff Concentrations

COD Concentration (mg/L)	Source	Location	Reference
27-290	Residential	Fresno, USA	FMFCD 1984
31-1,400	Residential	Fresno, USA	FMFCD 1984
12-460	Commercial	Fresno, USA	FMFCD 1984
150-2580	Industrial	Fresno, USA	FMFCD 1984
100	Freeways	U.S. National Average	Pitt 1985
174	Residential	Toronto, Canada	Pitt and McLean 1986
174	Commercial	Toronto, Canada	Pitt and McLean 1986
322	Industrial	Toronto, Canada	Pitt and McLean 1986

2.1.5 Deicing Agents

The primary deicing agent, NaCl, corrodes vehicles and bridges, contaminates drinking water supplies, and is toxic to many species of plants, fish, and other aquatic organisms (Amrhein et al. 1992; Brown 1994; National Research Council 1991). It can be found at concentrations ranging from 0.30 to 25,000 mg/l (Makepeace et al. 1995). Application of deicing salts, mainly NaCl and $MgCl_2$, during winter months is the main contribution to elevated chloride levels in snowmelt. An estimated 10 million tons of salt are applied to US roadways annually. Chloride is also introduced into stormwater runoff by tire road ballast, dust control, chemical manufacturing, wastewater treatment, fertilizers and insecticides (Makepeace et al. 1995). Chlorides can effect the environment directly by chloride toxicity, as well as indirectly by increasing the mobility of metals (Marsalek et al. 2003; Amrhein et al. 1992). Chloride adversely affects soil fertility by affecting soil structure and water transport through the soil (Marsalek et al. 2003).

Salinity variations in stormwater may affect plant growth rates by interacting with metal uptake, both directly and through the indirect effects of Na^+ and Cl^- ions. When grown in the absence of sediment, concentration and accumulation of Cu, Zn, and Cd decreases with increasing salinity in certain species of plants (Fritioff et al. 2005). However, under conditions where sediment is present, sediment may complicate the salinity effect due to the formation of colloid-sodium compounds that release metals from the soil/sediment and elevate the concentrations of metal available to plant roots and in the water column (Boukhars and Rada 2000; Greger et al. 1998). Salinity is not expected to be a concern in this application due to the fact that the test plots are located in a temperate location where deicing salts are not heavily applied.

2.1.6 Flow Rates

The two characteristics which are relevant to the quantity of runoff from a highway surface generated by a storm event are the intensity of the storm and the volume of the storm-derived runoff. The intensity of the storm can have a marked impact on the type and quantity of pollutants in runoff. This is due in large part to the fact that many pollutants are associated with particles, which are more easily mobilized in high intensity storms. A high intensity, short duration storm may have a higher pollutant load than a longer duration, less intense storm that generate the same volume of water. The runoff volume seems to have little effect on pollutant concentrations but is important in determining the total load to the receiving water. The volume of runoff that results from the directly connected impervious area is about equal to the volume of water from the rainfall. Rainfall excess results from both directly connected impervious areas and from the previous areas where rainfall intensity exceeds infiltration rate or the soil becomes saturated.

2.2 Current Stormwater Treatment Technologies

Historically stormwater treatment along highways has focused on the quantity of runoff as opposed to the quality of runoff because highway drainage is an essential part of roadway design because water collected on paved surfaces can pose a risk of hydroplaning or the freezing of water over the road surface, creating unsafe driving conditions (Czernick 2000). More recently, stormwater quality has become an essential element of highway design. Treatment systems for stormwater are constantly evolving as better and more cost effective alternative methods are discovered and evaluated. Differences in local conditions may result in the use of different stormwater treatment systems to achieve the same degree of pollution prevention and removal. The strategy of selecting an appropriate stormwater treatment method is considered as a Best Management Practice (BMP).

Various structural and non structural BMPs have been developed and used extensively for effective control of runoff flows and somewhat less effective control of stormwater quality (Viklander et al. 2003). Structural treatment methods are used for the removal of contaminants once they are in the system. Nonstructural treatment methods are designed to prevent contaminants from entering the system in the first place, and they include street cleaning, product substitution and controlled application of herbicides (US EPA 2002).

BMPs currently used to treat stormwater including but not limited to: detention basins, sand filters, grass swales, bioinlets, bioretention areas, hydrodynamic devices, infiltration trenches, porous pavements, wetland basins and media filters (Minton 2002). Although these methods are effective, they are often costly, require large areas of land for operation and/or fail to reduce metals in the effluent to safe levels (Minton 2002; Volesky 2003). Furthermore, many BMPs using vegetation in swales for treatment and flow control were based on model predictions developed for warm-weather climates subject to summertime rainfall events. In the Pacific Northwest, this is generally not the case as most runoff is generated during fall/winter months after a prolonged dry season. Cold climates also present additional challenges to the selection, design, and maintenance of stormwater treatment BMPs due to cold temperatures and dormant or near-dormant vegetation. Due to the limitations and restrictions of the techniques discussed above, there is a need to explore innovative, efficient, and cost-effective alternative technologies for the management of stormwater.

The filtration of urban stormwater runoff using sand and other media has been gaining acceptance as a structural best management practice for stormwater (Urbonas 1999). Overall filtration performance depends on many factors such as the desired treatment rate, use with other controls, the source water quality (types and concentrations of pollutants), and the physical characteristics of the media (type, size distribution, depth, and hydraulic loading rate) (Clark and Pitt 1999). Filtration has been emerging as a promising stormwater BMP especially with high removal rates for many of the key pollutants from highway runoff. Urbonas' work has led to reliable predictions for service life and filter maintenance scheduling for particulate removal, however, much work still needs to be completed in the area of dissolved pollutant removal by stormwater filters to predict bed longevity and maintenance procedures for the bed.

The use of both compost and vegetation as stormwater treatment methods is described in subsequent sections.

2.3 Application of Compost as Stormwater Treatment

Stormwater filtration is a relatively new use for compost and has been gaining popularity for use as a filter media. Compost can come from a variety of sources including food wastes, sewage wastes, industrial wastes and landscaping wastes. Composts made from yard waste, primarily leaves, have been found to have a very high capacity for adsorbing metals, oils, greases, nutrients, and organic toxins due to the humic content of the compost. Furthermore, compost has the capacity to beneficially act as a pH buffer (Ho et al. 2008) as well as erosion control along sloped roadsides (Persyn 2003). This section will review previous research on using compost to treat dissolved metals in stormwater as well as detail the mechanisms of how that is accomplished.

2.3.1 Compost Filters as Stormwater Treatment

Compost stormwater filters (CSFs), work by percolating stormwater through compost, which traps particulates and adsorbs dissolved materials such as metals and nutrients. Floating surface scums along with oil and grease are also removed. After filtering through the compost media, the filtered water can be channeled into a collection pipe or discharged into an open channel drainage way. Compost filters act as mechanical filters to remove fine sediments, as ion exchangers to remove solubilized ionic pollutants such as metals, as molecular absorption sites to remove organics, and provide biological substrate to aid in microbial degradation of organic compounds such as oil and grease (Clark and Pitt 1999). Stormwater is redirected via piping to the CSF where it then flows into a forebay before proceeding into the compost filter. The compost filter itself is confined with wood baffling around the perimeter as well as baffles within the compost filter which act as obstacles within the flow path in order to cause the stormwater to rise and use more of the compost filter. CSFs are located on top of drainage rock which allows stormwater to peculate into the ground and exit the system. The size of CSFs vary are dictated by expected stormwater flow rates. Drawbacks to CSFs are maintenance costs as well as the tendency for sediments to accumulate on the filter surface and affecting the infiltration capacity of the filter. The first generation composted leaf filter (proprietary deciduous leaf compost mixture) was first developed by W&H Pacific (now Stormwater Management, Inc.) for stormwater treatment (Johnson et al. 2003).

A three year study on the composted leaf filter showed excellent removal of metals and hydrocarbons (CSF Systems 1994). The compost filter is now sold in pre-packaged cartridge systems by Stormwater Management, Inc. Reported first flush compost filter removal efficiencies reported from W&H Pacific vary (Table 7).

Table 7- W&H Pacific Compost First Flush Efficiencies (CSF Systems, 1994)

Pollutant	Removal Rate (%)
Turbidity	86
Total Solids	63
Total Suspended Solids	94
Setteable Soilds	98
Total Volatile Suspended Solids	97
COD	79
Total Phosphorus	63
Ammonia	65
Total Kjeldahl Nitrogen	72
Copper	83
Zinc	86
Lead	86
Aluminum	88
Iron	93

Since the introduction of the first generation compost leaf filter, several similar technologies have emerged including the use of yard waste compost contained in mesh under the patent of Fiterexx Filter Soxx (Faucette 2004).

2.3.2 Compost as a Roadside Stormwater Treatment

The use of compost filters as discussed above requires that stormwater be redirected to a point of treatment which is costly, maintenance intensive and can result in high flow rates. As an alternative to compost filters, compost blankets can be applied along roadsides. The advantages are that no special structure is required and the treatment system can be placed along isolated stretches of highway, where implementation of traditional stormwater treatment methods is not feasible. Successful use of compost blankets to control erosion and sediment control has been demonstrated for a number of years in the United States (Persyn 2003; Faucette 2004). These applications are governed by the American Association of State Highway and Transportation Officials MP-10-03 standard (AASHTO Designation MP-10-03, 2005). Specifications for compost used in erosion control include particle size, moisture content, organic matter content, pH, soluble salt content, and human-made inert contents.

Recent studies completed at state government test sites and universities have also demonstrated that elements, namely humics, within compost have the ability to filter, bind and degrade contaminants from stormwater passing through it (Grimes et al. 1999). (Johnson et al. 2003) observed that compost placed within horizontal beds was able to reduce Zn, Cu, Pb within the stormwater passing over it by 86%, 83% and 86% respectively for the first year of operation. Compost was found to have excellent chemical and physical properties for the sorption of dissolved metal ions (Cu^{2+}, Pb^{2+} and Zn^{2+}). Batch sorption data were used to determine the sorption efficiency of Cu (93%), Zn (88%) and Pb (97%) by compost (Seelsaen et al. 2007).

Research indicates that metabolically inactive biomass, such as compost, is capable of binding metals even at low concentrations comparable to those found in runoff (Grimes et al. 1999). Using compost to treat metals in stormwater has strong advantages namely, a competitive performance, metal selectivity, low cost, and potential regeneration and metal recovery (Volesky 2003). The main advantage is that is can treat large volumes of runoff with a low concentration of metals (Gavrilesku 2004).

Compost filters possess a high buffering capacity in the alkaline range. When the influent is between pH 6.7 and 8.3, the effluent is consistently between pH 7.0 and 8.0. In addition to binding dissolved metals, humic compounds are capable of buffering pH. Humic acids may benefit plant growth by chelating unavailable nutrients (Mackowiak et al. 2001).

2.3.3 Mechanisms Involved in Metal Binding by Compost; Ion Exchange and Sorption

Yard waste compost has the capacity to bind metals with binding sites on the surface and throughout the material itself (Volesky 2003). Humic substances, the largest component of organic matter in compost, are the primary species involved in sorption. The process of metal binding involves the transfer of dissolved solids (sorbates) from water to a media filter (sorbent) in this case; compost (Minton 2002). Humic substances have been known to reduce metal toxicity by binding the metal (Grimes et al. 1999). Physiorption and chemisorption metal binding mechanisms have been proposed to be active in compost with adsorption and ion exchange respectively (Grimes et al 1999).

In physical adsorption, molecules and ions bind to solid surfaces by weak Vander Waals or electrokinetic forces (Minton 2002). The sorbent media utilizes the concentration gradient between the sorbent and sorbate (Volesky 2003). There is no significant redistribution of electron density in either the molecule or at the substrate surface. The adsorption process follows both the Freundlich and Langmuir isotherms with pore diffusion being only one of the rate-controlling steps (Namasivayam and Periasamy 1993). In chemisorption, a chemical bond, involving substantial rearrangement of electron density, is formed between the sorbate and sorbent. During ion exchange, the preferred ion in the fluid replaces the less preferred ion in the material (Minton 2002). The inclination for ion exchange is related to the electrical charge of the ion and commonly, the preferred ion replaces a cation or anion with a lower charge (Minton 2002). This process continues until all the binding sites are taken up by the preferred ions and this point the material is considered to be exhausted (Minton 2002). Factors affecting the rate of adsorption include surface area of the sorbent, nature of sorbate, and surface tension of the sorbate.

Humic substances are typically the major component of organic matter in compost and the major species involved in sorption. Chemically, humic acid contains more carbon and less hydrogen than does the animal and plant residues from which it has formed through extensive biological decomposition (Tipping 2002). Classically, humic substances are classified into three classes based upon their solubility in alkaline or acidic conditions. Humin is the fraction not soluble in water at any pH value; humic acid is not soluble under acidic conditions (pH <2) but becomes soluble at higher pH; and fulvic acid is soluble at all pH conditions (Tipping 2002). Metals and other cations may become complexed by humic substances. Chelation by neighboring carboxyl and phenolic groups is generally considered to be the major mode of metal complexation (Tipping 2002). Metals in solution may also bind with dissolved organic matter or with suspended particulate matter (Minton 2002). Binding with the dissolved organic has been found to out compete sorption to suspended particulate matter, resulting in a greater overall dissolved metal concentrations (Tipping 2002). However, even though dissolved organics increase the overall dissolved metal concentration, they do not necessarily increase the toxicity to aquatic organisms as the metal hydrates are generally not toxic (Tipping 2002). Moreover, results from studies indicate that the metals in compost are tightly bound to the compost matrix and are not easily removed unless harsh conditions are used (Grimes et al. 1999).

There are several physical benefits of using humic acids as a soil media including an increase the water holding capacity, increased aeration, improved seed bed and reduced erosion. Chemical benefits of humic acids include an increased buffering capacity, the ability to chelate metal ions under alkaline conditions, providing organic and mineral substances essential to plant growth, retains water soluble inorganic fertilizers in the root zones and releases them to plants when needed, possesses extremely high ion-exchange capacities and promotes the conversion of a number of elements into forms available to plants. Figure 1 shows a general rendition of a humic acid molecule. Appendix B contains additional background information on humic substances.

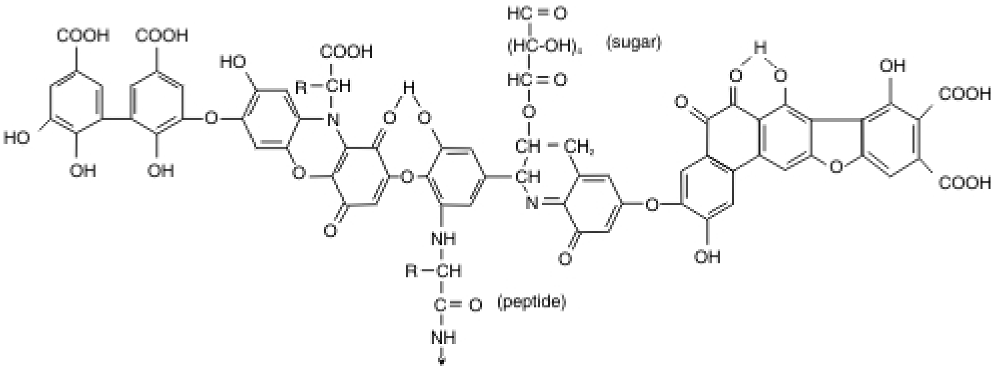


Figure 1- General Chemical Structure of Humic Acid

2.3.4 Limitations of Compost as a Stormwater Treatment Method

One challenge in using compost as a stormwater treatment method is the large variability that results from different feed stocks, processing technologies, and product screening techniques which can produce composted organics with diverse physical, chemical, and plant growth characteristics. Consistent compost properties are often difficult to achieve (pH, humics content, particle size, etc) however they can be quantified and standardized with appropriate efforts.

A secondary limitation of using compost in stormwater treatment is that ions are 'leached off' whenever a pollutant sorbs to the media because compost is an ion-exchange resin. Soluble phosphorus is a common ion that is leached off during ion exchange (influent, 0.09 - 1.0 mg/L; effluent, 0.29 mg/L) (CSF Systems 1994). Soluble phosphorus likely is released from the captured solids through microbial action and since the compost only has a weak anion exchange capacity, most of the soluble phosphorus is not removed from the water once it is leached from the compost. Testing has also shown an increase in boron and nitrate in the effluent of the compost filter (CSF Systems 1994). Other areas of concern in sorption treatment beds, in general, relate to the capacity of a bed to remove metals through precipitation and filtration, breakthrough instability of metal precipitates.

A final limitation to consider is that turbidity and colour are often higher in the effluent as a result of the organic acids in compost. When the filter goes dry between storms, color producing organics are likely released from the medium and retained in the pores, waiting to be washed out during the next filtration. The minimum expected effluent concentrations for the composts and filter are 10 mg/L for suspended solids, 100 HACH color units, and 10 NTU for turbidity which will occur regularly for most stormwater filters (Clark and Pitt 1999). Effluent turbidity is a parameter that is necessary to monitor while conducting laboratory or field testing of compost as a stormwater treatment method.

2.4 Roadside Application of Vegetation as Stormwater Treatment

Vegetative stormwater BMPs include such technologies as grassed swales and vegetated roadside filter strips. Vegetated filter strips, also known as buffer strips are relatively smooth vegetated areas with moderate slopes that accept stormwater runoff as overland sheet flow. This BMP is most closely related to the use of roadside composted/vegetated system that is examined in this report. The mechanisms of pollutant removal for vegetated strips are filtration by grass blades (or other vegetation), sedimentation, adsorption, infiltration into the soil, and biological and chemical activity in the grass/soil media. Although not originally designed specifically for water quality treatment and often not recognized by regulatory agencies, vegetated areas, such as medians and shoulders have the potential to provide substantial pollutant reduction under specific conditions. Factors that affect the removal efficiency of vegetated BMPs treating urban runoff include vegetation type, slope, flow velocity, flow depth, season, and length.

Traditional vegetated BMPs rely on actively growing plant life as an integral part of the pollutant removal processes (Kaighn Jr. and Yu 1996; Han et al. 2005). When (some) plant species go dormant at the onset of colder temperatures the overall pollutant removal efficiency of the system is reduced (Ouellet-Plamondon et al. 2006). Due to the temporal nature of runoff in the GVRD, design storms generally occur prior to the establishment of vegetation or after the growing season under near-dormant conditions. Because of this, additional research is required before to using dormant vegetation in stormwater treatment. The following section will review previous research and identify limitations of using vegetative strips to treat dissolved metals in stormwater as well as detail the mechanisms of how that is accomplished.

2.4.1 Previous Research on Vegetative Strips as Stormwater Treatment

Vegetated swales and filter strips have not always been accepted as primary controls for the treatment of stormwater runoff. This is mainly the result of the wide range of pollutant removals reported for vegetative controls in various studies (Schueler et al. 1992; Young 1996). Consequently, a lack of confidence emerged among regulatory agencies that vegetative controls could provide reliable and consistent removal of pollutants in stormwater. Some design manuals recommend vegetative controls only for pre-treatment to reduce sediment loading to filtration systems or other structural stormwater controls. Many of the studies in which lower removal efficiencies were observed were not well designed and significant removal of the pollutants occurred before the runoff entered the test sections that were monitored (Caltrans 2003).

To date, most historical highway runoff studies were conducted in the field using built highways because highway environments such as traffic, precipitation, etc. are very difficult to physically model in a laboratory. The historical field studies that evaluated the performance of vegetated roadsides can be categorized into three types. The first type analyzes the soils from the roadsides adjacent to the highways to determine whether the soils act like a "sink" to retain pollutants (Bell and Wanielista 1979). The second type collects samples from concentrated flows in swales and compares them with untreated runoff samples from pavements (Yousef et al. 1985; Kaighn Jr. and Yu 1996; Barrett et al. 1998). The third type collects samples by intercepting overland flows on grassy shoulders before they reach the swale and compares samples collected at different distances from the edge of pavement (Barrett et al. 2004). Regardless of the sampling type, vegetated roadsides are recognized to have positive nutrient removal performance (Kaighn and Yu 1996; Barrett et al. 1998, 2004). The remaining issue on the performance of vegetated roadsides is that the range of removal performance is widespread (Barrett et al. 2004). Table 8 below summarizes previous studies findings on pollutant removal efficiency:

Table 8- Summary of Previous Filter Strip Studies

Study	Removal Efficiencies	Notes
Schueler et al. (1992)	28% TSS	Recommended velocity <0.76 m/s, length>15 m.
Kaighn and Yu (1996)	64% TSS; 59% COD;-21% TP; 88% zinc	Specifically highway runoff.
Young (1996)	70% TSS, 40% P, Zn; 25% Pb;10% NO3/NO2	Efficiencies from cited study.
Barrett et al. (1997)	85% for TSS; 68%-93% for turbidity, chemical oxygen demand (COD), zinc, and iron; 36%-61% for total organic carbon (TOC), nitrate, total Kjeldahl nitrogen (TKN), total phosphorus (TP), and lead; and negative removal of bacteria.	Relatively short (7-9 m) filter strips with moderate slopes (9%-12%).
Line and Hunt (2008)	Load reduction efficiencies in all pollutants ranging from 24 to 83% and the highest reduction for TSS.	The largest negative efficiency was for nitrate+nitrite nitrogen (NO2+3–N). The increase in NO2+3–N likely resulted from a combination of nitrogen additions within the cell and conversion of other forms of nitrogen to NO2+3–N.

(Glick et al. 1993) investigated the effect of vegetative cover and several other factors on filter strip effectiveness in an urban area. Four different vegetated covers were compared: wooded areas, wooded areas cleared, native unmowed grasses, and native mowed grasses. The forested areas produced the highest concentrations of pollutants, while the mowed and unmowed areas generally had the lowest concentrations. Overall, grassed areas were found to be more effective at removing pollutants than forested areas. In addition, vegetative composition was found to have a significant impact on filter strip effectiveness. In addition to vegetation type, vegetation density is also a factor in pollutant removal. (Yousef et al. 1985) examined a thick grass cover (80% grass, 20% bare soil) and found it to have reduced nutrient removal efficiencies when compared with a thin grass cover (20% grass, 80% bare soil). This finding was attributed to increased decay of organic matter where thick grass cover was available.

(Li et al. 2008) compared the removal efficiencies of stormwater pollutants of two distinct highways where vegetation was present. The research was able to determine specific removal efficiencies but a control plot without vegetation was not available making the argument that the vegetation was the mechanism and not the soil matrix difficult. More experimentation is required in order to validate and quantify the true effects of the vegetation.

2.4.2 The Effect of Vegetation on Subsurface Flow and Erosion

Slope instability and erosion of the soil by water and wind are major environmental hazards which can result in ecological or economical damages. Vegetation has proven to be an effective protective buffer between atmospheric elements and soil and is one of the most commonly used erosion control practices along highways. Vegetation reduces erosion both above and below ground by slowing the velocity of runoff, stabilizing the slope, and stabilizing accumulated sediment in the root zone of the plants. Roots are able to stabilize the soil in two different ways.

First, roots and root remnants physically bind soil particles and in this way form mechanical barriers for soil and water movement (Tengbeh 1993).The mechanical influence of roots is controlled by several parameters: diameter, degree of bifurcation, appearance of root hairs, friction between root and soil and, root network distribution (Abe and Ziemer 1991). Shallow interlocking root networks can contribute to the mechanical reinforcement of soil media, acting as an anchored net of densely interwoven roots (Sidle et al. 1985). Dense root mats carpet the ground and provide soil cohesion, which ultimately limits erosion by overland flow (Sidorchuk and Grigorev 1998). However, living and dead root systems can provide subsurface water flow pathways by creating biopores and thus reducing the amount of erosive overland flow, which may contribute to channeling. By penetrating the soil mass with roots, the soil is reinforced bringing about an increase in cohesion and hence, in soil shear strength (Waldron and Dakessian 1981; Morgan and Rickson 1995). Secondly, roots and root remnants excrete binding agents and form a food source for micro-organisms that in turn produce other organic bindings (Reid and Goss 1987). These bindings increase the amount of stable soil aggregates in the long term and thus reduce soil erodibility (Hartman and De Boodt 1974).

In addition to erosion reduction properties, plants can also affect soil moisture distributions and the soil hydraulic properties either directly by root water uptake (Feddes et al. 1988; Zhuang et al. 2001) as well as by accumulating water inside the root biomass, or more indirectly by modifying the soil pore structure through the growing root system (Angers & Caron 1998; Kodesova et al. 2006). Vegetation density also has an effect on the performance of vegetated roadsides. When roadsides are densely covered with grasses above 90%, significant sediment removal is expected, often within the first four meters of the edge of pavement. Dormant (in winter conditions, not actively growing) vegetation barriers are as effective as active barriers for reducing runoff (>10%) and sediment (>90%) (Blanco-Canqui et al. 2004).

2.4.3 Mechanisms Involved in Metal Binding by Vegetation; Bioaccumulation and Biosorption

The mechanisms and toxicity information regarding metals and their effects on plants are important to consider during the development of stormwater treatment applications especially when vegetation is part of the system. Plant species have the capacity to uptake metals for different reasons like sequestration, draught resistance, disposal by leaf abscission, interference with other plants and pathogen/ herbivore defense (Boyd 1998). Many plant species are capable of accumulating more than 1000 mg/g dry leaf weight and are termed hyperaccumulator plants (Brooks et al. 1977). However, phytoremediation that takes place as a plant is actively growing is not an element of this research as the vegetation is considered dormant under the climate conditions presented. The following section gives an overview of the mechanisms used by vegetation to bind metals as well as the potential toxic effects.

Bioaccumulation is the active mode of metal accumulation by living cells and is dependent on the metabolic activity of the cell (Yang et al. 2005). Metal transport is a developing field in plant biology with many aspects still unknown**.** Metal tolerance may reflect the ability of an organism to survive in an environment with a high concentration of metal or to accumulate a high concentration of metal without dying. Alternatively, metal tolerance may be due to a species ability to prevent the intracellular buildup of metal concentrations toxic to its metabolic processes (Yang et al. 2005).

The plant root represents the first barrier to the selective accumulation of ions present in the soil solution. Uptake and kinetic data for the nutrient ions and chemically related non-nutrient analogs suggest that metabolic processes associated with root absorption of nutrients regulate both the affinity and the rate of absorption of specific non nutrient ions (Cataldo and Wildung 1978). Different mechanism or mechanisms are characteristic to each species of vegetation and are sensitive to abiotic environmental factors. Plants possess a range of mechanisms to uptake and potentially store dissolved metals are generally more tolerant to the presence of metals in soil. The effects of specific metal species on vegetation are discussed in detail in Appendix C. Mechanisms potentially involved in the uptake of metals include: (i) binding to the cell wall; (ii) reduced uptake or efflux pumping of metals at the plasma membrane; (iii) chelation of the metal in the cytosol by various ligands, such as phytochelatins, metallothioneins, and metal-binding proteins; (iv) repair of stress-damaged proteins; and (v) the compartmentalization of metals in the vacuole by tonoplast-located transporters (Hall and Williams 2003).

In soil and soil solutions, plants are exposed to many elements simultaneously and to other factors which can interact in an infinite number of ways. Thus, the integrated effects of multiple metals may be quite different from obtained when evaluating and individual factor or element. A basic condition for normal development of plants is a chemical balance of elements in the soil and plants. The conditions may vary due to species, genotypes, age of plant and other factors. Interactions between elements do occur naturally in solids and plants. The effects can be antagonistic, additive or synergistic. Although interactions between metals are commonly observed, they are often complex and contradictory (Pahlsson 1989). Thus, interaction effects may be inconsistent, depending on the parameter measured in the plant, plant species and concentrations of metals.

Lethal metal toxicity is not expected to be a concern in this application. The concentrations of dissolved metals in stormwater are low, and the application is sporadic. Moreover, Zinc and Copper are essential elements to higher plants and are involved in several metabolic processes whereas lead and cadmium are not known to have any function in plants (Pahlsson 1989). It is important to note that although numerous investigations have been made on the toxic effects of metals on higher order plants, the conditions are different than those proposed in this document. Many of the toxicology studies discussed in Appendix C involve submersion of vegetation in an aqueous concentration of metals whereas this application would involve contact with the dissolved metals in stormwater in a short-term, intermit basis. The metal concentrations used are often so high that they are not a realistic representation of ecological conditions. Frequently investigations have been made in traditional liquid cultures where the stated concentration pertains to the start of the experiment or immediately after each replacement of the solution. Further, consideration is not taken to the metal ion activity in the soil or nutrient solution not to chelated elements or interactions with other ions. The experimental periods are also short in duration ranging from a few hours to a few weeks. This complicates the possibilities of assessing whether a longer period of exposure to lower metal concentrations in the plant medium causes growth disorders (Pahlsson 1989).

Appendix C details the toxic effects of Zn, Cu, Cd, Pb, Cr, and Fe on vegetation. This information was used to ascertain if plants could be resistant or tolerant to elements at concentrations commonly found in stormwater runoff and insight as to species of plants that were most suitable for use along side roadways under these conditions.

2.5 Relevant Literature Concerning Modeling of Sorption and Flow through Porous Media

Relevant literature pertaining to sorption breakout curves, used to model metal sorption in media and residence time distributions used to model hydrodynamic conditions in reactors are detailed in this section.

2.5.1 Sorption Breakthrough Curves

The dissolved metal sorption capacity associated with composted and composted/vegetated beds can be evaluated from chemical breakthrough curves. Breakthrough curves can be generated by passing a stream of dissolved metal solution through the treatment bed until all of the binding sites in the material have been exhausted and the bed is considered chemically expired. As explained in Section 2.3.3, dissolved metals are capable of being removed from stormwater through sorption and ion absorption. As the sorbate flows through the bed, the dissolved metals are removed from the liquid phase and simultaneously, the solid phase gradually saturates its binding sites starting at the inlet (Geankoplis 1993). Once the entire bed gradually saturates with metals, the effluent ion concentration will increase to its influent concentration. The effluent concentration is plotted against time to produce a representative breakthrough curve of the specific system.

As sorption proceeds, the inlet concentration of the ion in both the sorbate and the sorbent change with time and with position in the bed. Initially, the sorbent concentration is assumed to be zero and therefore when the fluid first contacts the inlet of the bed, most of the adsorption occurs here. In theory, shortly after flow begins, the solid near the entrance is becomes saturated and most of the adsorption now takes place at a point slightly farther from the inlet. Later, at t_2, the mass transfer zone where most of the adsorption occurs has moved down the bed. As a result, the concentration of metal in the sorbate remains constant down the mass transfer zone where it begins to drop. As the flow continuous, the mass-transfer zone (S shaped curve), moves down the column. Ideally, the effluent concentration remains at zero until the mass transfer zone starts to reach the column outlet. Once the mass transfer zone reaches the column outlet, the effluent concentration begins to increase until it reaches the breakout point, c_b (Geankoplis 1993). Figure 2 below shows the breakthrough concentration profile of the effluent from the bed.

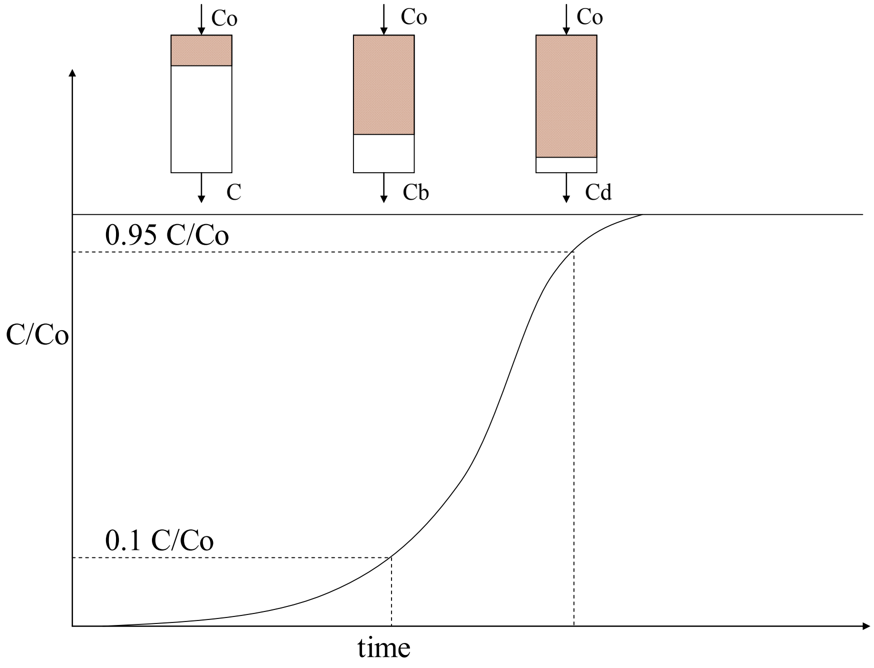


Figure 2- Adsorption Zone Progression in a Fixed Bed Adsorber (Co=initial influent concentration, C=concentration of effluent)

The velocity at which the mass transfer zone moves down the column depends on sorbate loading, sorbent capacity, and the column feed rate. Ideally, the shape of the mass transfer zone remains constant as it travels down the column; however, axial and radial mixing cause deviations from the ideal piston flow resulting in the flattening of the breakthrough curve. After this, the concentration rises rapidly up to c_d, which is the end of the breakout curve where the bed is judged ineffective. The break-point concentration is often taken as 0.01 to 0.05 for c_b/c_d. The value for c_d is generally taken to be approximately equal to the inlet concentration, c_0 (Geankoplis 1993). The process of metal ion sorption is governed by three steps: the sorption equilibrium, the sorption particle mass transfer, and the flow pattern through the packed bed. The overall effect of these steps determines the overall performance of the packed bed which in turn is judged by the breakpoint (Volesky 2003).

2.5.2 Residence Time Distribution Curves

Residence time distribution (RTD) theory provides a useful tool for analyzing non-ideal flows (Levenspiel 1972; Kadlec 1994). The RTD of a reactor is a probability distribution function that describes the amount of time a fluid element could spend inside the reactor, (as mentioned earlier, the reactor in this research refers to the treatment bed). The time the atoms have spent in the reactor is called the residence time or retention time of the atoms in the reactor. RTDs are used to characterize the mixing and flow within reactors and to compare the behavior of real reactors to their ideal models. Deviation from ideal behaviour may be the result of dispersion RTD measurements provide an effective technique to diagnose flow behaviour within a wide range of flow systems and can reveal flow distribution characteristics such as transit times, short-circuiting, re-circulation zones, and dead zones. The unique hydrodynamic characteristics associated with composted and composted/vegetated beds can be determined experimentally using tracer tests to produce RTD curves at different flow rates. In an ideal plug-flow reactor (PFR), all the atoms of material leaving the reactor have been inside it for exactly the same amount of time. A real reactor may have non-uniform flow patterns that do not conform to the ideal PFR mixing patterns because of corners, baffles, non-uniform media packings, etc.

Three assumptions generally govern the theory of residence time distributions:

1. The reactor is at steady-state.
2. Transports at the inlet and the outlet takes place only by advection.
3. The fluid is incompressible.

RTD curves can be generated with the introduction of a conservative tracer to the system with the two most common methods of injection being pulse input and step input. The scope of this report is limited to step input with the methods and analysis from Elements of Chemical Reaction Engineering (Fogler 2005). In a step input, the concentration of tracer at the reactor inlet changes abruptly from 0 to C_0. The inlet concentration, $C_{0,}$ is kept constant until the concentration in the effluent is indistinguishable from that of the influent (ie. when C(t)= C_0 the test is complete). A concentration versus time graph, C(t), gives the time distribution of tracer concentration which is used to generate the subsequent exit-age distribution curves, E(t), and non-dimensional curves, F(t). The concentration of tracer at the outlet can be measured and normalized to the concentration C_0 to obtain the cumulative distribution function, F(t):

$$F(t) = \left[\frac{C_{out}}{C_0}\right]_{step} = \frac{C(t) - C_o}{C_f - C_o} \qquad [1]$$

$C(t)$ effluent concentration at time t;
C_o effluent concentration at time zero; and
$C_f$concentration of the feed solution.

Fluid elements may require differing lengths of time to travel through the reactor. The distribution of the exit times, defined as the E(t) curve, is the RTD of the fluid. E(t) quantitatively describes how much time different fluid elements have spent in a reactor. It is the most used of the distribution functions because it characterizes the length of time various atoms spend at reactor conditions. The exit concentration of a tracer species C(t) can be used to define E(t). From F(t), the exit age E(t) can be found:

$$E(t) = \frac{d}{dt}\left[\frac{C(t)}{C_0}\right]_{step} = \frac{dF(t)}{dt} \qquad [2]$$

By knowing the E(t)-curve, the mean residence time can be obtained. The residence time (τ) or hydraulic retention time is equal to the mean residence time (t_m) and is a measure of the average length of time that a soluable compound remains in a reactor:

$$t_m = \frac{\int_0^\infty tE(t)dt}{\int_0^\infty E(t)dt} = \int_0^\infty tE(t)dt = \tau \qquad [3]$$

These can be related to the volume and the volumetric flow rate *v* in the mathematical relation

$$\tau = \frac{V}{\upsilon} = \int_0^\infty tE(t)dt \qquad [4]$$

3.0 Objective

After reviewing the above literature, additional information on sorption capacity and hydrodynamic properties needs to be explored before it can be determined if the use of compost and vegetation filter strips is a viable option for the treatment of roadside stormwater runoff for the removal of metals. The overall purpose of this paper is to advance the knowledge of metal sorption and flow mechanisms in roadside vegetated compost and compost treatment beds in order to substantiate the future use of this application as a feasible dissolved metal treatment alternative. Given that metals persist in the environment, any stormwater BMP for control of metals and particulates are essentially temporary repositories, and as such they must be replaced or regenerated based on loadings, sorptive breakthrough behavior, and effluent objectives. As a result, media capacity and breakthrough behavior for sorptive filter media are critical factors with respect to media specifications.

The overall objective of the research is to determine differences in flow pattern and metal sorption capacity between compost and vegetated compost roadside strips that receive metal loadings during the cooler, non-growth period. The sub objectives for this research are as follows:

1. To generate residence time distribution curves for compost and vegetated composted filters at a variety of runoff flow rates and obtain retention times.
2. To generate chemical breakthrough curves in order to assess the metal sorption performance of vegetated compost and compost filter strips at a temperature of 11°C.
3. To investigate the effluent pH and turbidity of vegetated compost and compost filters over time when a constant metal loading is applied as influent.
4. To make inferences on practical application based on the observed data

4.0 Materials and Methods

This section details the materials and methods selected in order to ascertain the thesis objectives.

4.1 Materials and Experimental Apparatus

This section details the species of vegetation and compost sorbent media selected to complement the climate, precipitation patterns, flow rates and metal concentrations common to the immediate application area or the GVRD. The apparatus used for experimentation as well as the choices made for the stormwater and tracer influents are also discussed.

4.1.1 Vegetation Selection

As mentioned above, the selection of plants was based on their compatibility with climate conditions, soils, and topography and their ability to tolerate urban stresses from pollutants, variable soil moisture, and water levels (Han et al. 2005). The seeding mix and ratio for the selected vegetation of interest was taken from Section 757 of the British Columbia Ministry of Transportation (BC MOT) Regulations typically used for roadside application. The choice of species for GVRD was the Vancouver Island/ Coast Mix and is detailed below in Table 9. This vegetation was selected because of its compatibility with the biotic and abiotic environment present in the area of interest and its capability to germinate on compost. Additionally, it preserves the integrity of the native ecosystem by introducing typical roadside non-invasive vegetation in a controlled manner.

Table 9- BC MOT Vancouver Island/Coast Mix

Species	Percent by Weight
Perennial Ryegrass	26%
Creeping Red Fescue	24%
Alsike Clover	14%
Hard Fescue	13%
White Clover	9%
Timothy	8%
Canada Bluegrass	4%
Redtop	2%

The guidelines outlined by the BC MOT were followed to ensure optimum growth. The seed ratio was determined from using the mean annual precipitation as a parameter. In the Lower Mainland of British Columbia the annual precipitation is greater than 90 cm (See Appendix A for consultants report and historical rainfall data). There for the corresponding seeding density for hydraulic application is recommended at a rate of 50-80 kg/ha and the application of fertilizer is recommended at a ratio of 12-32-6 (NPK). Cultivation of the vegetation took place in the laboratory media bed (see section 4.1.3 for detailed description of bed). 18.95 kg of Filterexx grow media was placed in the beds and the seeding mix described above was distributed according to BC MOT guidelines. The seeds germinated in 20°C temperature and the vegetative grass was allowed to mature for six months (length of time to simulate spring and summer growing season) (Harper-Lore and Wilson 2000). Vegetated beds were then gradually acclimatized over one month to an 11°C environment similar to typical winter temperatures in the GVRD. Watering of the beds was done weekly and the beds were fertilized every two weeks with Miracle-Gro grass fertilizer (NPK: 12-32-6). Vegetation was grown for seven months in order to allow it to become established and mature before experimental trials were conducted. The amount of roots present at maturity was quantified using the ratio of root mass to compost mass. Appendix D outlines the procedure followed to ascertain the root ratio.

The conditions that the vegetation was exposed to during experimentation were chosen in order to ensure that the vegetation was dormant and not metabolically active. This was done in order to minimize the number of variables and concentrate on the structural properties of the vegetation only. No nutrients or light were supplemented during the experiments to decrease metabolic activity. Using the vegetation as structural contribution only simulated the 'worst-case scenario' if the vegetation was not active at all during the winter months.

4.1.2 Sorbent Media Selection

The sorption media chosen for the experiments was Filterexx Growing Media yard waste compost from Filtrexx International. As shown in the literature review, well-aged yard waste compost has been utilized to prevent erosion along highways (US EPA 1997), and has the capacity to bind metals and filter particulate matter (see section 2.3

Sorption). Filterexx Grow Media was chosen because it is certified to meet the standard specifications recommended by the US-EPA in section 503. Additionally this compost meets the particle size and nutritional needs to allow vegetation to germinate and establish. (Risse and Faucette 2001) recommended a mixture ratio of 3:1 (fine, coarse) for optimum vegetation growth and cover. Fine particle grade refers to 0.00635 to 0.0127 m screened and coarse grade 0.0508-0.0762 m. Filterexx Growing Media meets these requirements with a mean particle size of 0.0063 m and a particle distribution of ratio of 3:1 (fine, course) as shown in Figure 3. Additional parameters for Filterexx Grow media were determined as follows: bulk porosity of 0.267 (+/- 0.05, n=3, 1 standard deviation), dry weight bulk density of 0.332g cm–3 (+/- 11.75, n=3, 1 standard deviation) and moisture content of 0.502 (+/-0.048, n=3, 1 standard deviation). Experimental procedures for these parameters are outlined in Appendix D.

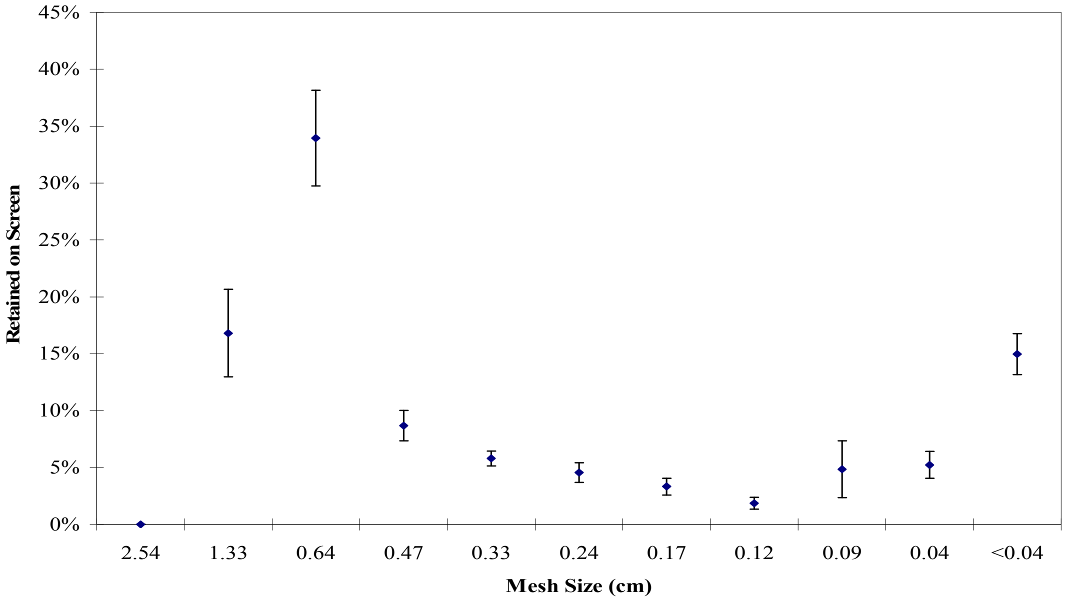


Figure 3- Particle Distribution of Filterexx Yard Waste Compost Growing Media

The depth of compost required for growth on road embankment is 0.0508 m (US EPA 1997) and 0.1016 m for erosion control (US EPA 1997). Therefore, a depth of 0.1016 m (4.0 inches) was used as the depth in treatment beds in order to address both vegetation establishment as well as erosion control. Additionally, material and bed slope 12 ° (see section 4.1.3) conform to BC MOT construction and maintenance branch design build standard specifications.

4.1.3 Experimental Apparatus and Flow Distributor

An experimental bed was designed and constructed to conduct both the runoff simulations and the bromide tracer studies. The bed was constructed to represent actual field conditions while maintaining precise laboratory controls. When filled with compost, the bed became the packed-bed reactor used in both sorption and flow experiments. The bed is a rectangular plexi-glass box with no top and dimensions 1.02, 0.254, and 0.10 m of length, width, and height, respectively, with wall thickness of 0.00635 m. Two different flow distributors were used in conjunction with the experimental bed. Bromide tracer studies used a 7-weir flow distributor to ensure an even distribution of runoff (see Appendix E for full specifications) and a compost bed length of 0.65 m (long bed) in order to minimize potential edge effects and channeling (**Error! Reference source not found.**). Zinc sorption studies used a cylindrical flow distributor to simulate rainfall and a shorter compost bed length 0.3 m (Figure 5). Complete design drawings for both the experimental bed and flow distributors can be found in Appendix E.

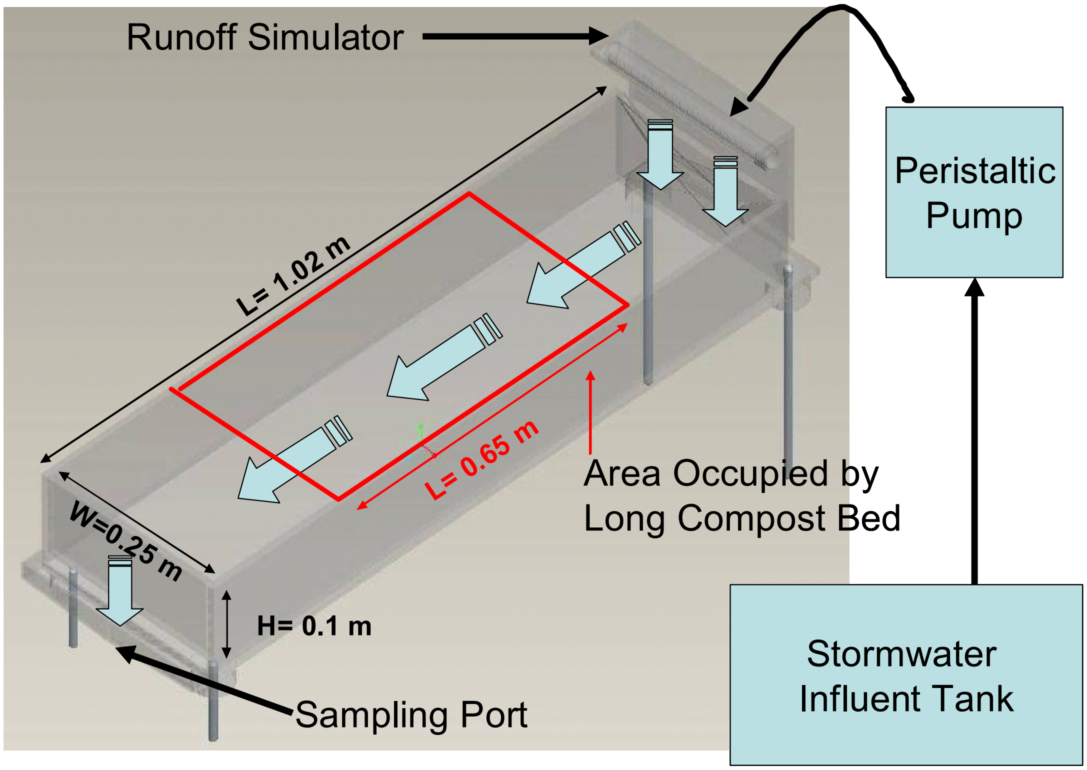


Figure 4- Experimental Bed with Flow Distributor (bed slope=12°) Long Bed Configuration for Bromide Tracer Experiments

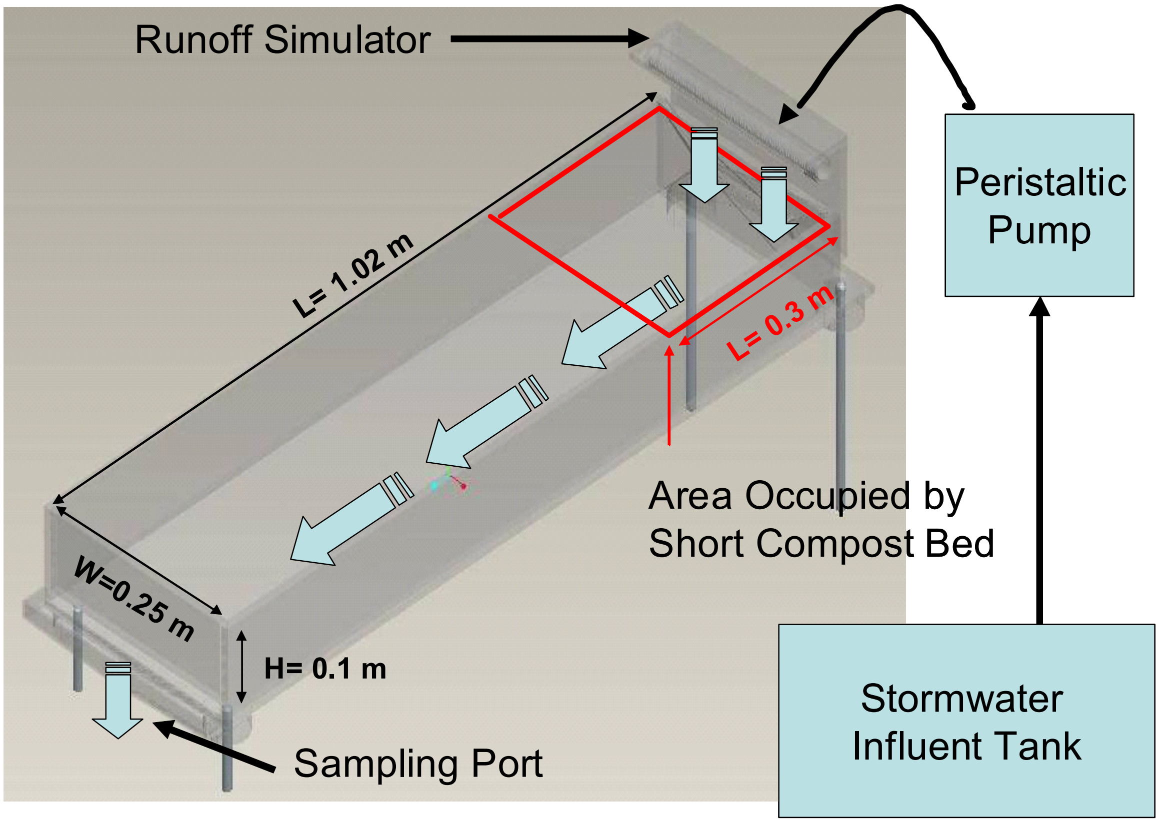


Figure 5- Experimental Bed with Flow Distributor (bed slope=12°) Short Bed Configuration for Zinc Capacity Experiments

The bed was set on a slope of approximately 12° to mimic the slope of the roadside and allow the simulated runoff to move through the bed via gravitational flow. The bromide tracer solution and mock runoff solution are described in sections 4.1.4 and 4.1.5. Not shown in the illustration are Cole Parmer (Vernon Hills, Ill) Masterflex pumps and clear Tygon tubing which were used in conjunction with the experimental bed.

4.1.4 Flow Rates

Mass balance methods are used to equate rainfall intensity to runoff rate. For a completely impervious watershed, the volume of rainfall excess is equal to the volume of precipitation. Precipitation intensity onto a watershed of contributing area must equal the instantaneous rate of discharge from the watershed using appropriate conversion factors and sufficient travel time for the total area to contribute runoff. The rational method can be used for computing peak flows on urban and rural watersheds and relates the runoff rate to the catchment area and rainfall intensity to calculate the volumetric rate of runoff generated during a rainstorm. The rational method can be described as follows:

$$Q = \frac{CiA}{360} \text{ (Metric)} \qquad [5]$$

Where,

$Q = Run\ off\ \ flowrate\ (L^3 / t);$

$C = Runoff\ \ coefficient;$

$A = Catchment\ area\ (L^2);$

and

$i = Intensity\ of\ \ the\ rain\ (L / t).$

Since rainfall intensity is rarely constant over time, average intensity is used in Equation 5. A runoff coefficient of 0.95 is used for impervious surfaces (MOAFRA 2009).

For this research, representative runoff rates for storm events in Lower Mainland of British Columbia were calculate for a runoff catchment area pertaining to two lanes of the four lane busy road, W 16th ave located at the University of British Columbia (UBC). The runoff from the catchment area is expected to enter into a 1 m wide compost roadside strip. The catchment area was estimated to be 18.1 m^2 by Urban Systems Ltd using historical rainfall data for UBC (See Appendix A for consultants report and historical rainfall data). The flow rates 70, 264, 676 and 900 mL/min were calculated to correspond to low flow, 1-year, 2-year and 5 year storm events representative of the GVRD.

4.1.5 Tracer Selection and Preparation

Bromide has been shown to be a conservative tracer for soil-water in batch sorption experiments (Levy and Chambers 1987). Bromide was chosen as the ideal tracer because it is a nonreactive species that is easily detectable. Bromide has physical properties similar to those of the reacting mixture (water) and is completely soluble. The selected tracer did not modify the physical characteristics of the fluid (density, viscosity) and the introduction of the tracer did not modify the hydrodynamic conditions of the system. Batch sorption experiments that were conducted using the methodology from "Bromide as a Conservative Tracer in Soil-Water Studies" (Levy and Chambers 1987) showed that bromide did not adsorb on the walls or other surfaces of the reactor or bind with the vegetation or compost (see Appendix F).

The bromide stock solution of 1.0 mol/L was prepared by weighing a mass of 119.0 g potassium bromide (molar weight 119.0024 g/mol**)** in an analytical balance. The potassium bromide was transferred into a 1 L Nalgene flask with distilled water. The solution was then continuously mixed with a magnetic stir bar. In order to create the 0.2 mol/L required for the bromide tracer experiments, 1.0 L of this stock solution was added to a 5.0 L tank filled with distilled water. The tank was manually stirred for several minutes in order to minimize a concentration gradient in the tank. Sample calculations for obtaining the targeted bromide concentrations can be found in Appendix F.

4.1.6 Choice of Metal and Concentration

The influent stormwater used in the sorption trials was restricted to single-metal trials although multi-ions are usually present in runoff (see Section 2.1.2 Metals). Zinc was used because it is the most mobile of the metals commonly present in stormwater (Burton and Pitt 2002) and therefore would yield the most conservative estimate of sorption. The selection of zinc allows the opportunity to examine the most mobile of the metals found in stormwater and gives a 'worst-case scenario' profile for metal sorption. Had a less mobile metal such as nickel or aluminum been selected, the sorption capacity generated would not identify the high end of the range of metal mobility. Zinc is also economical to measure using standard methods. The zinc concentration used was approximately 3.0 mg/L, which is two orders of magnitude greater than the B.C Aquatic Guidelines for zinc concentration (see Table 3) but be under the World Health Organization legal limit of 5.0 mg/L for discharge purposes. The use of a higher concentration shortened the sorption capacity experiments to a reasonable length of time as well as allowed for economical on site testing with use of the Smart 2 Colorimeter.

The zinc stock solution of 5.0 g/L was prepared by weighing a mass of zinc chloride in an analytical balance. The zinc chloride was transferred into an acid-washed

1.0 L Nalgene flask with distilled water. Nitric acid was added to the zinc solutions in order to ensure the zinc chloride was completely dissolved. The solution was then continuously mixed with a magnetic stir bar. In order to create the 3.0 mg/L required for the zinc sorption experiments, 118.2 mL of this stock solution was added to a 197.0 L acid-washed tank filled with tap water. An aeration pump was submerged in the tank for continuous mixing in order to minimize the zinc concentration gradient in the tank. The pH of the tank was adjusted to 5.6 using sodium hydroxide to target pH of rain water. Rainfall is naturally acidic (pH 5.6) due to formation of carbonic acids from the carbon dioxide in the atmosphere (Novotny 1995). The pH measurements were made using a pH probe (WTW pH 330i). Sample calculations for obtaining the targeted zinc concentrations can be found in Appendix F.

4.2 Methodology and Analytical Techniques

A total of three experiments were performed in order to address the objectives outlined in Section 3.0. First, a flow characterization study was conducted using bromide tracer, followed by zinc sorption capacity experiments and finally the pore volume of the treatment beds was quantified at different flow rates. The general methodology will be described, followed by details pertaining to specific analysis. Finally, statistical methods used to evaluate results are outlined.

4.2.1 Hydraulic Flow Characterization Experiments

In order to determine the effect of vegetation on the subsurface flow of the treatment system, the residence time distribution (RTD) of two different treatment bed configurations were examined:

- Filterexx growing media- compost only
- Filterexx growing media with mature (7 month) grass vegetation- vegetated compost

This experiment simulated transport of a non-reactive chemical tracer in a laboratory scale model of a roadside treatment bed. Four flow rates (see Table 10) pertaining to typical storm events in the GVRD. All experiment trials were performed as in triplicate on a slope of 12° in the laboratory at 21°C.

Table 10- Bromide Tracer Study Flow Rates

Storm Event	Flow Rate (mL/min)
Low flow	70
1-Year	264
2-Year	676
5-Year	900

The RTD was determined by introducing a conservative tracer into inlet of the treatment bed at time t=0 and measuring the tracer concentration, C, in the effluent stream as a function of time (see Figure 6). Prior to the introduction of the tracer, water was run through the inlet port of the bed at the desired flow rate for 1 hour in order to ensure steady-state flow. This application served three purposes: to saturate all preferential pathways of flow through the treatment bed, to remove any remaining residual tracer from the previous run and to ensure that each trial took place with compost under similar moisture conditions. Flow rate was measured frequently at the outlet of the bed using a stopwatch and a graduated cylinder over the duration of the experiment.

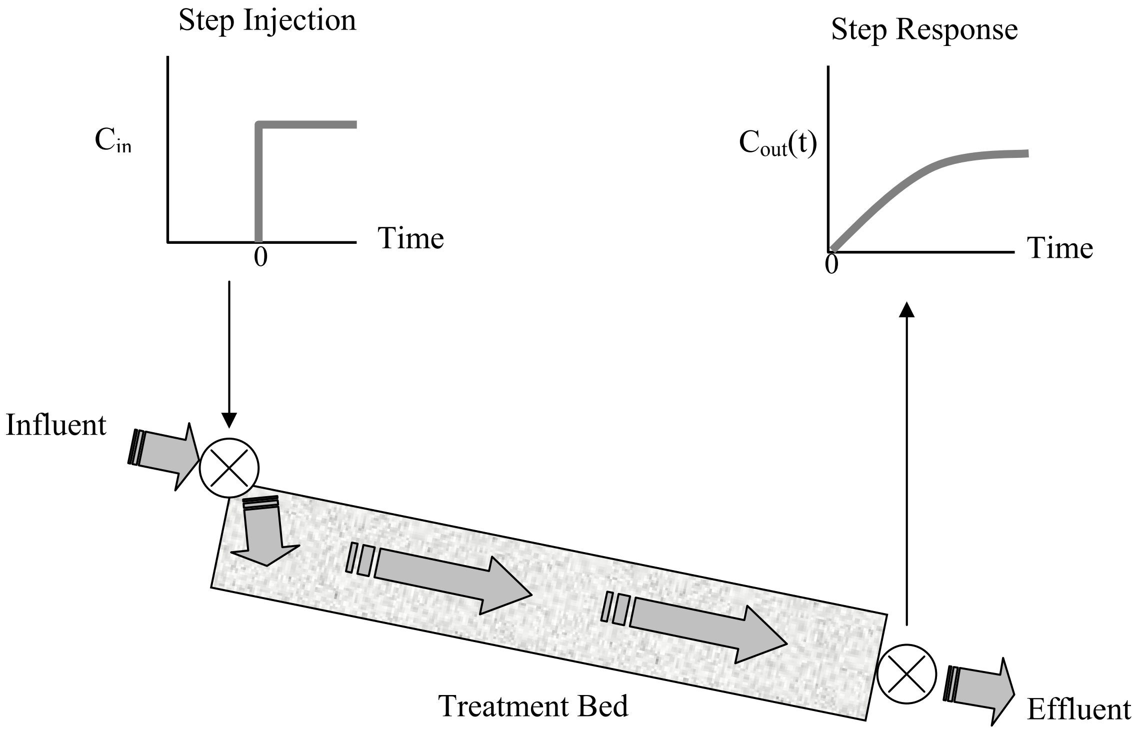


Figure 6- Bromide Tracer Schematic with step change at t=0; $C_{t=0}$ =0.0 M Br^-, $C_{t>0}$=0.2 M Br^-

The step change was induced by transferring the Masterflex tubing from a water tank into a tank containing 0.2 M of potassium bromide (KBr). An air bubble was formed in the tubing at the time that the feed tank was switched, which was used to determine time zero for each experiment. Effluent samples were collected from the treatment bed outlet in 10 second intervals with the exception of the low-flow rate which was collected in 30 second intervals due to volume constraints. The sampling ceased when the concentration of the effluent equalled the concentration of the influent in order to achieve a true step-change.

The effluent samples were analyzed using a Fischer Scientific Accumet Bromide Combination Ion Selective Electrode (13-620-524) and a Fischer Scientific Accumet Excel XL 40 Series Meter which relates electronic conductivity to Br- concentration. Calibration curves were generated by preparing bromide solutions with concentrations in the rage of 10^{-5} to 1.0 mol/L and measuring their electronic conductivity. A linear relationship with a correlation coefficient [R] of 0.991 was obtained using the same stock

KBr solution used for experiments. The Bromide Ion Selective Probe was calibrated before each experimental trial analysis and representative calibration curves can be found in Appendix F. From each sample bottle, 5.0 mL of effluent was extracted and 0.1 mL of Ionic Strength Adjustor (ISA) (5.0 M Sodium Nitrate, get manufacturer) was added to a glass tube. Additionally, bromide tracer studies were conducted on the apparatus only to obtain a baseline curve of the equipment, pump and tubing at each flow rate and are presented in Appendix F.

Data collected from each tracer event were transformed into RTD curves for normalized comparison. The normalized RTD curves where found using the procedure from Wastewater Treatment (Tchobanoglous et al. 2003) and Elements of Chemical Reaction Engineering (Fogler 2005). Results from tracer tests were used to compute C(t) graphs for each data set. The data was then transformed into the cumulative distribution function F(t) defined in Equation 1. From the cumulative distribution function, F(t), the exit age E(t) was found using Equation 2. Finally, the mean residence time (hydraulic retention time) t_m was calculated as using Equation 3.

The free water volume of the treatment beds was determined using the free air space (FAS) according to standard protocols specified by the US Composting Council (TMECC 2002). The free water volume is a measurement of the available pore space during a specific flow rate. The free water volume varies with flow rate and has a distinct wetting front (interface between soil that is unchanged from the initial state and the newly wetted zone from an infiltration or irrigation event). The flow path and pooling also vary between flow rates and treatment beds. Figure 7 illustrates the various water phenomena observed and measured during flow characterization experiments.

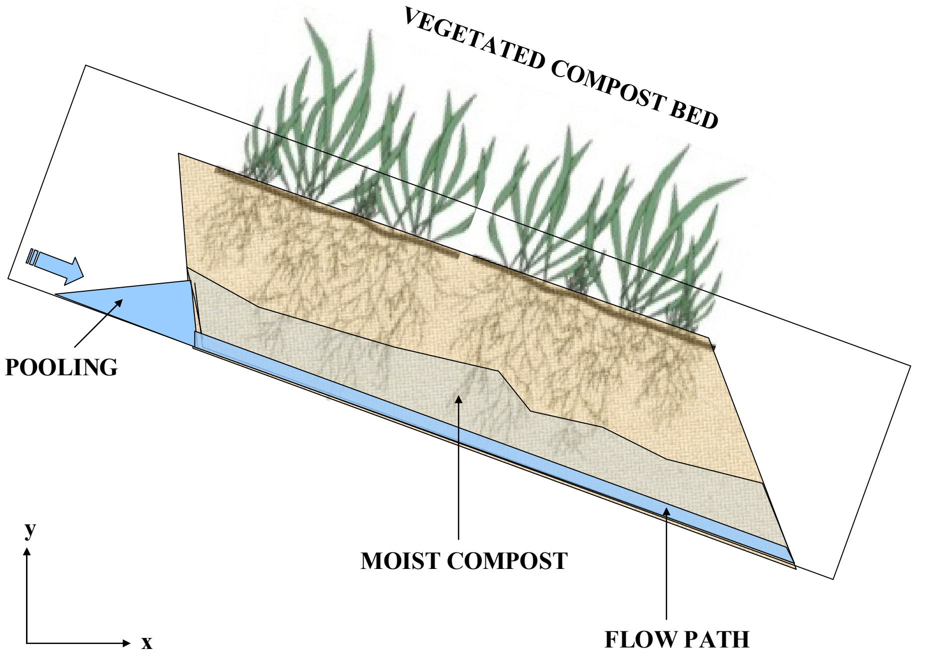


Figure 7- Defining the Wetting Boundaries of System

In order to compare the water volume usage of the treatment beds, experiments were conducted to determine the free water volume (Vf) for the treatment bed configurations at different flow rates. The treatment beds were operated at the flow rate of interest with deionized water for 1 hour to ensure equilibrium was reached. Flow rate was measured at the outlet of the bed using a stopwatch and a graduated cylinder during the duration of the experiment. Any pooling visible in the bed was measured and calculated to be Vp. At time t=0 the liquid supply to the treatment bed was ceased and liquid draining from the bed was collected for 30 minutes. The volume of this liquid was measured to determine the total draining volume (Vt). Vf was determined by the following equation:

$$Vf=Vt-Vp \qquad [6]$$

The porosity and bulk density of the compost was used to calculate the total pore volume (Vp). Then, the ratio of free water volume to total volume of the bed was calculated from Vf/Vp with units being mL/mL.

4.2.2 Sorption Capacity Experiments

This experiment investigates the metal sorption of a laboratory scale model of a roadside treatment bed over time. In order to determine the effect of vegetation on the dissolved metal sorption capacity of the treatment system, two different packed-bed reactor configurations were examined:

- Filterexx growing media- compost only
- Filterexx growing media with mature (7 month) grass vegetation- vegetated compost

A metal sorption profile was determined by introducing zinc solution into inlet of the treatment bed at time t=0 and measuring the tracer concentration, C, in the effluent stream as a function of time. A constant flow rate of 3.0 mg/L of zinc at a pH of 5.6 and a flow rate of 264 ml/min until chemical breakout occurred and the beds were exhausted. Although carefully prepared and calculated, the incoming zinc concentration of the runoff simulations varied from 0.27 to 0.39 mg/L over the length of the experiment (see Appendix J for quality control table regarding zinc concentration) due to mixing. A Cole-Palmer peristaltic pump was used to apply the runoff influent to the beds and was calibrated frequently during use (see Appendix F). Flow rate was measured at the outlet of the bed using a stopwatch and a graduated cylinder frequently throughout the duration of the experiment. All experimental trials were performed in triplicate on a slope of 12° and kept at a constant temperature of 11°C. Due to a mechanical failure with the peristaltic pump, one replication was not completed.

During experimentation, effluent samples were collected in acid washed polyethylene bottles at regular intervals from each bed. Sampling frequency for Trial 1 was every hour; samples for Trial 2 were sampled and analyzed less frequently in order to conserve money and resources using the chemical breakout information obtained from Trial 1. The samples were analyzed for pH, turbidity and dissolved zinc. Turbidity and pH samples were measured using a La Motte Smart2 Colorimeter and a WTW portable pH meter with temperature sensor, respectively. Dissolved zinc concentration was determined using a LaMotte Zinc test kit and a LaMotte Smart2 Colorimeter. Before analysis, the samples were filtered through 0.45 µm acid-washed filters (Persyn 2003) using 20 mL acid-washed syringes. Additionally, zinc solutions with concentrations ranging from 0 to 2.5 mg/L were used to calibrate the Smart2 Colorimeter before each experimental trial. The calibration solutions were made using the same stock zinc solution used for the runoff simulation and representative calibration curves can be found in Appendix F. The procedure for the LaMotte Zinc test can be found in Appendix G.

4.2.3 Statistical Analysis

Turbidity and pH of compost only and vegetated compost beds were subjected to ANOVA one-way analysis of variance to test differences. All analyses were performed using Excel 2003.

5.0 Results

This chapter presents the results obtained from the experiments conducted. For all experiments, two treatment bed configurations were compared and contrasted; a compost-only bed and apparently dormant mature vegetated bed (7 months of growth from germination). During the first few months following germination, clover species dominated the vegetation beds. As the beds matured, rye species became more prominent. At maturity, the majority of the vegetation appeared to be rye grass. Prior to being exposed to experimental conditions, the vegetation appeared healthy and green. After the trials, the vegetation appeared yellow and dry. The composition of vegetative species was consistent over the course of the experiments. The ratio of root mass to compost present in the vegetated compost beds at 7 months was found to be 0.117g/g +/-0.025 (n=3, 1 standard deviation). The bottom image of Figure 8 shows the short (30 cm) vegetated compost beds used in the zinc sorption experiments. Longer beds (65cm) of the same root density and maturity were used in the bromide tracer experiments. The top image in Figure 8 shows the root matting present at the bottom of the bed where the compost was in contact with the plexiglass bed during germination and growth.

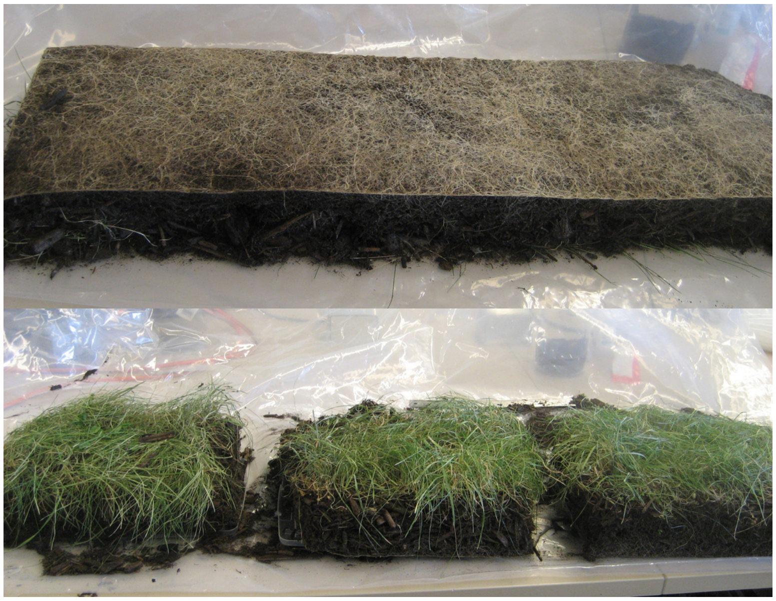


Figure 8- Mature Vegetation (7 months) Used in Experiments. Top image shows long vegetated strip bed (65cm in length) flipped over to illustrate root mass. Bottom image shows short vegetated compost strips (30cm in length)

5.1 Hydraulic Flow Characterization Experiments

The following section presents the results obtained from the experiments designed to quantify the hydraulic subsurface flow characteristics of the two treatment bed configurations.

5.1.1 Bromide Tracer Results

The measured bromide concentration of effluent was plotted versus time to compile C(t) curves (see Figure 9, Figure 10, Figure 11 and Figure 12 below). E(t) and F(t) curves derived from Equations [1] and [2] can be found in Appendix H. During the higher flow rates, 676 and 900 mL/min, both the vegetated compost and compost only beds experienced a backup of water before the bed which resulted in pooling (see images in Appendix L). The wetting front of the water on the saturated treatment beds was not measured but varied depending of apparent bed dryness from trial to trial. Figure 9 shows the slope of the C(t) curve for the compost bed carried out at 70 mL/min is steeper than that of the vegetated compost bed. Both curves exhibit a flattened ‘S’ shape. The curves are separate, distinct and do not overlap each other.

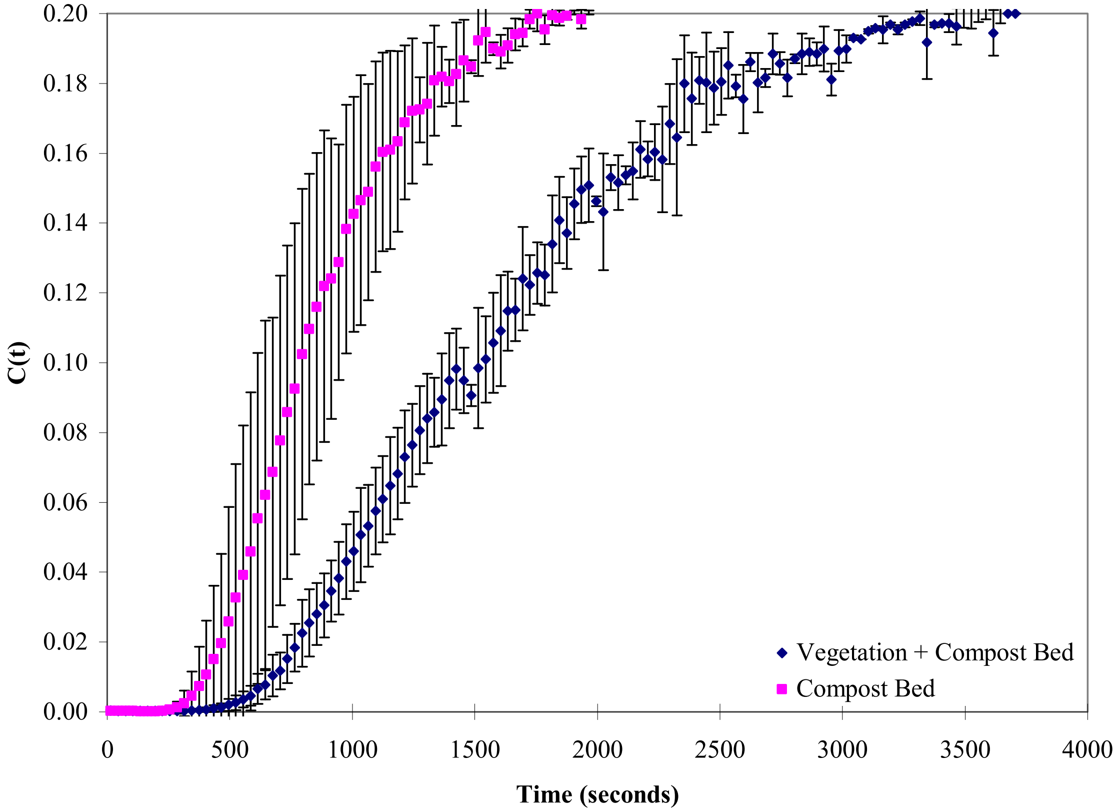


Figure 9- Low Flow, 70 ml/min Storm Event Concentration Curve, n=3, error bars= one standard deviation, bed length=65cm

Curves constructed from data obtained over the higher flow rates are, as evidenced by overlapping error bars, indistinguishable for the rest of the storm events and all tend to be ‘S’ shaped (Figure 10, Figure 11, Figure 12).

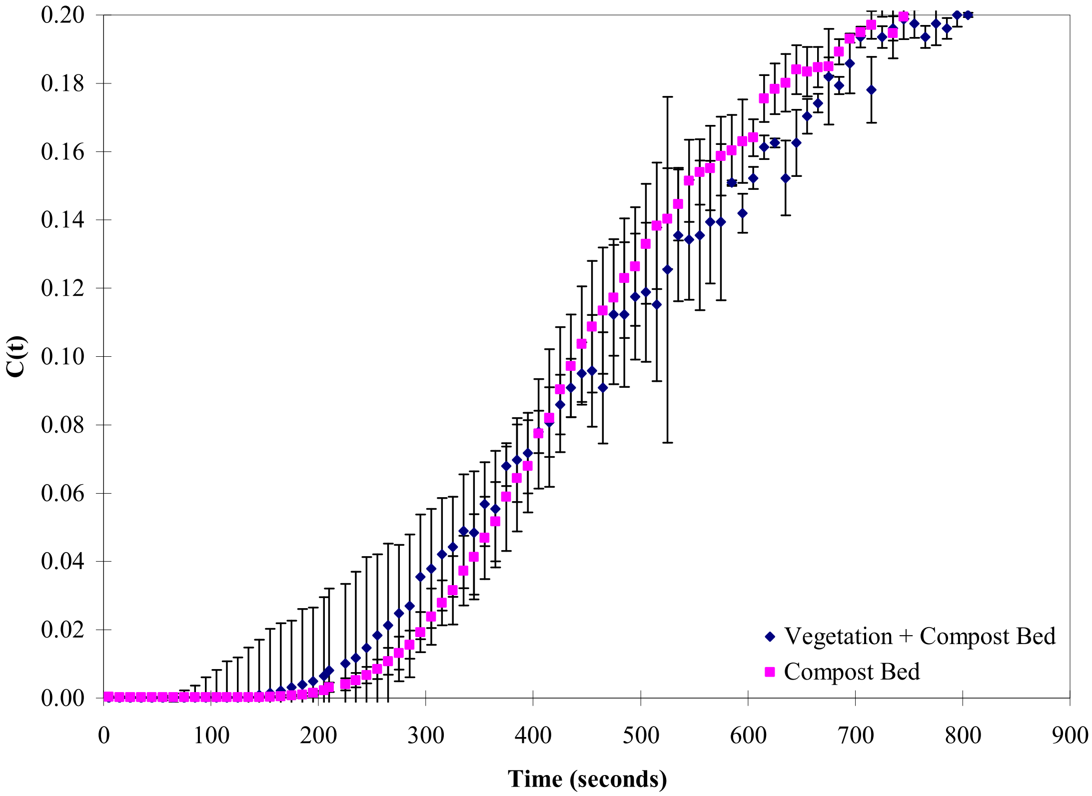


Figure 10- 1-year, 264 ml/min Storm Event Concentration Curve, n=3, error bars= one standard deviation, bed length=65cm

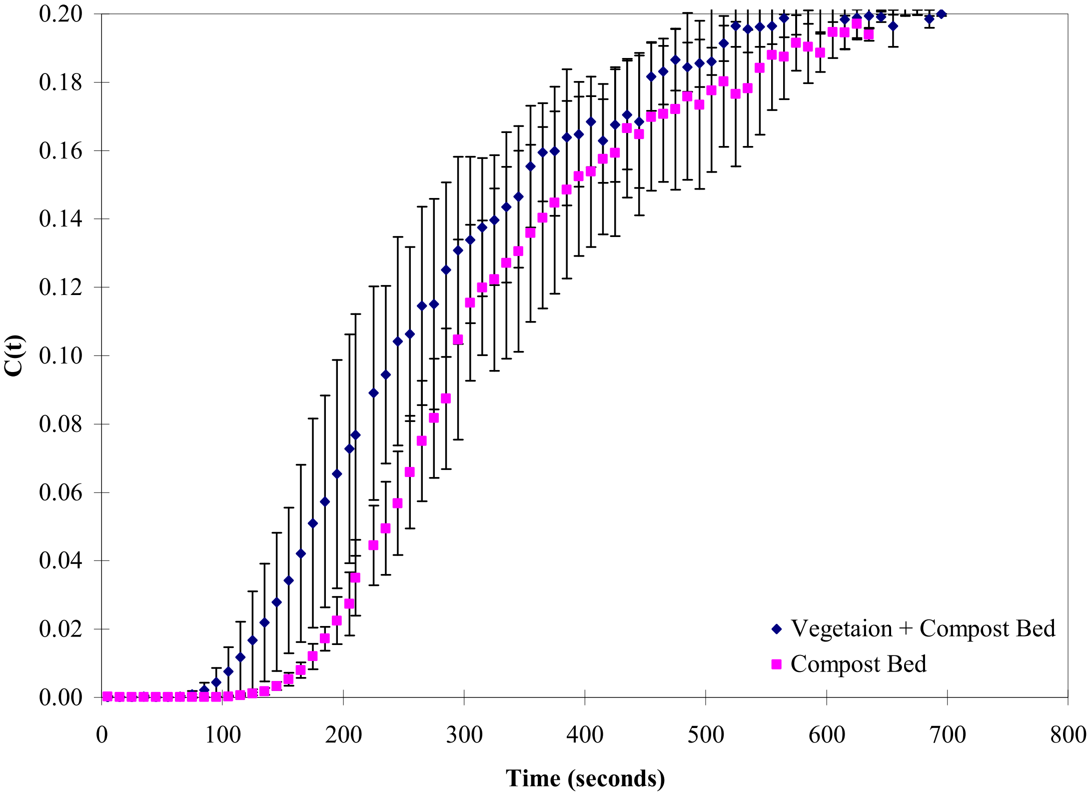


Figure 11- 2-year, 676 ml/min Storm Event Concentration Curve, n=3, error bars= one standard deviation, bed length=65cm

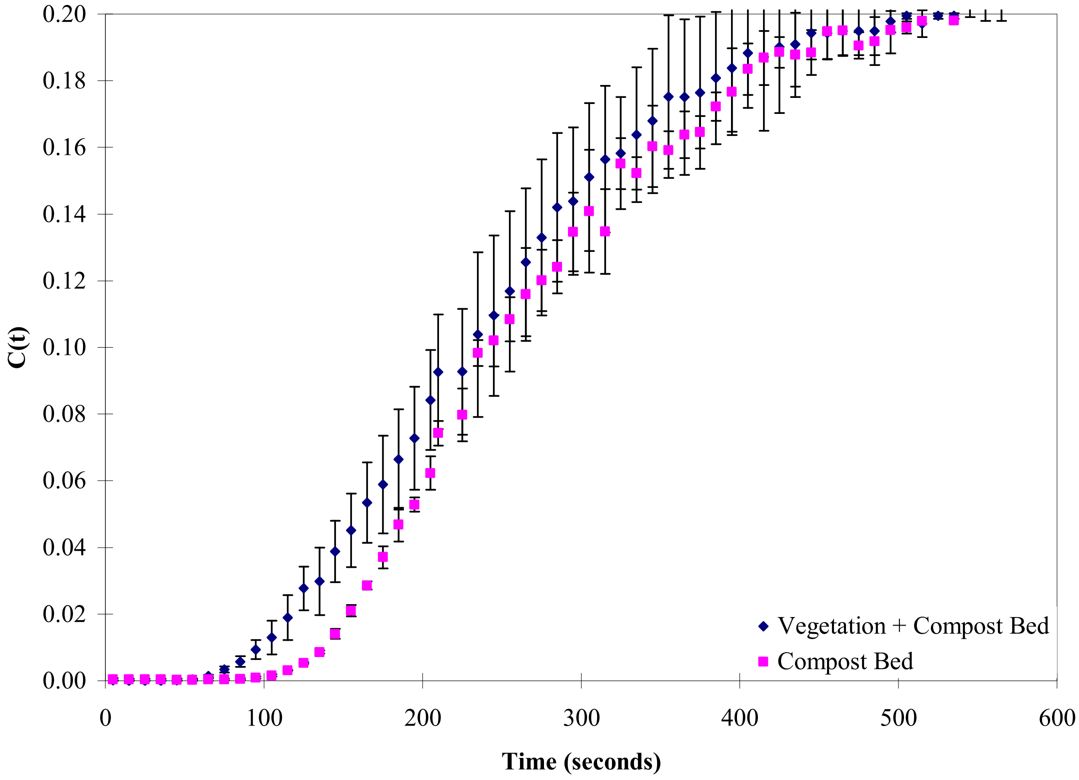


Figure 12- 5- year, 900 ml/min Storm Event Concentration Curve, n=3, error bars= one standard deviation, bed length=65cm

The retention time calculated from the graphs above represents the total retention time including the flow distribution apparatus. The retention time of the apparatus was also calculated for each of the flow rates (see Appendix F) were subtracted from the total retention times. The corrected retention times are corresponding to flow rates and bed configurations presented below in Table 11 and represent the amount of time an average molecule spends in the treatment bed.

Table 11- Retention Times

Storm Event	Vegetated Compost Bed Retention Time (seconds)	Compost Only Bed Retention Time (seconds)
Low flow storm event (70mL/min)	417.40	221.68
1-year storm event (264mL/min)	93.61	87.92
2-year storm event (676 mL/min)	85.96	75.31
5-year storm event (900 mL/min)	66.13	59.25

Overall, the retention times for the 264, 676, and 900 mL/min flow rates vary only slightly between the vegetated compost and compost only bed configurations with the longer retention times favoring the vegetated beds. The low flow event shows a significant difference between the vegetated and compost only configurations with the vegetated bed having a much longer retention time.

5.1.2 Effect of Length of Treatment Beds on Retention Time

In order to verify the effect of the length of treatment bed on retention time, two treatment bed configurations (compost-only bed and mature vegetated compost bed) were investigated at two different lengths: 30 cm and 65 cm. The ‘short’ bed configuration used the exhausted zinc sorption capacity beds (section 4.2.2) and compared them to the ‘long’ beds used for the original bromide tracer trials (section 4.2.1). Figure 13 below illustrates the RTD curves of both the vegetation and compost only treatment using different bed lengths.

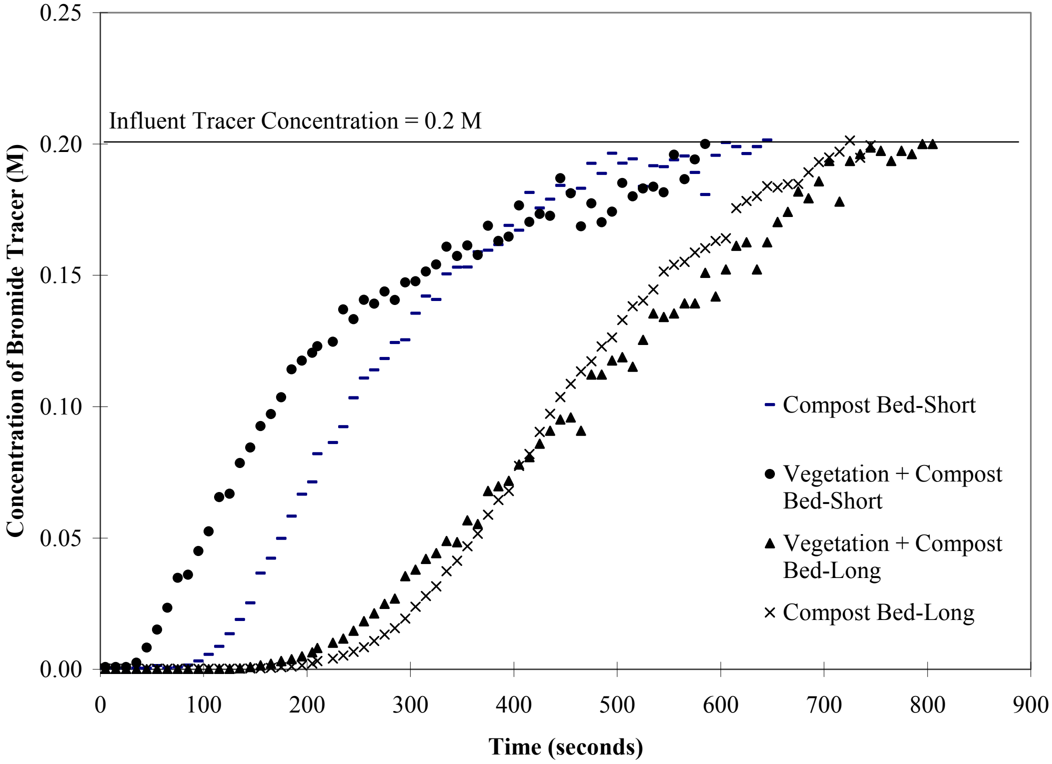


Figure 13- Comparison of Different Lengths of Treatment Beds Under a 1-year storm Event (264 mL/min), n=3, standard deviation error bars not shown for clarity purposes, long bed= 65cm, short bed=30cm

The retention time was calculated for each of the bed configurations and is presented below in Table 12:

Table 12- Effect of Length of Filter Bed on Retention Time (n=3, average shown)

Bed Length	Vegetated Compost Bed Retention Time (seconds)	Compost Only Bed Retention Time (seconds)
short bed- 30 cm	73.00	75.00
long bed- 65 cm	93.61	87.92

As would be expected, a longer treatment bed yields a longer retention time. An interesting observation is that the shape of the long beds and short beds is different. The long beds exhibit an 'S' shape curve and the short beds exhibit linear characteristics for the first 250 seconds before curving up to a final concentration of 0.2 M.

5.1.3 Free Water Volume Usage of Beds

In order to examine the effects subsurface flow on roadside filter strips the volume of free water present in the compost pores of the treatment bed configurations was determined at different flow rates and is presented in Figure 14. During the higher flow rates, 676 and 900 mL/min flow rates, both the vegetated compost and compost only beds experienced a backup of water before the bed which resulted in pooling (see images in Appendix L). The wetting front of the water on the treatment beds was not measured but varied considerably from trial to trial.

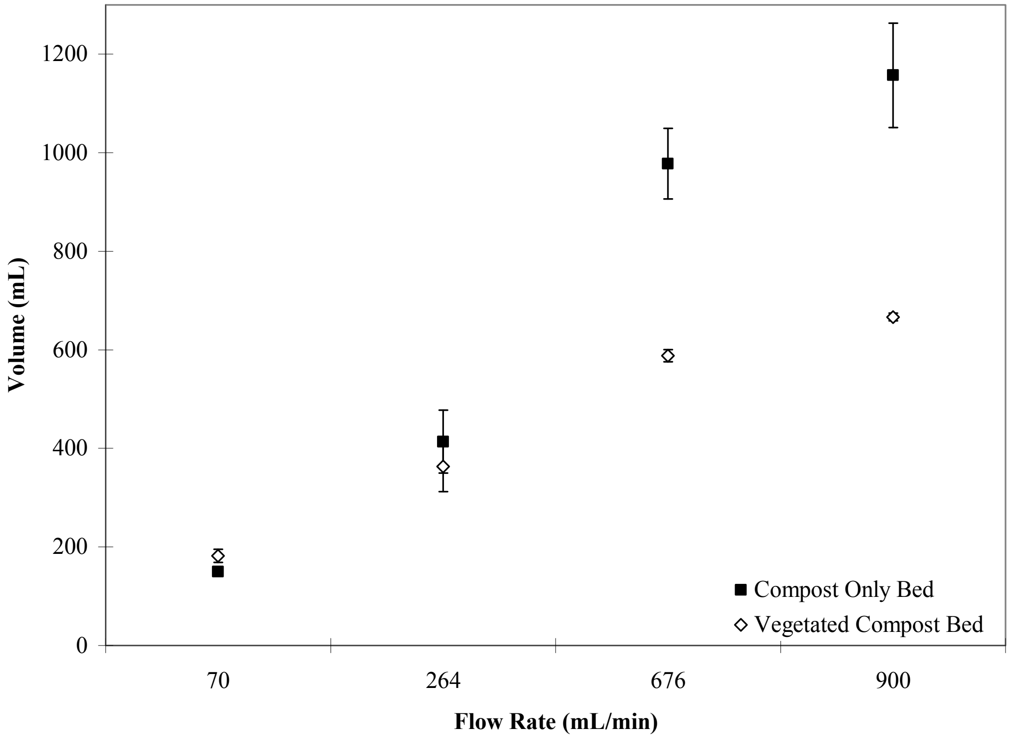


Figure 14- Free Water Volume Usage of Treatment Beds at Different Flow Rates (Slope=12°, n=3, error bars= one standard deviation)

During all experimental flow events, the entire bed was never fully saturated with flow. The flow ran along the bottom of the treatment beds without fully saturating the bed volume. Depending on flow rate, the wetting front and pooling behind the compost bed varied also occurred. The volume of water that pooled behind the treatment beds was quantified and the results are presented in Table 13.

Table 13- Volume of Water Pooling Behind Treatment Beds (n=3, bed length= 65cm, average value shown)

Flow Rate	Compost Only Free Water(mL)	Vegetated Compost Free Water (mL)
Low flow storm event (70mL/min)	0.00	0.00
1-year storm event (264mL/min)	1349.71	823.97
2-year storm event (676 mL/min)	1676.53	1131.65
5-year storm event (900 mL/min)	1846.11	1445.32

The volume of the treatment bed exposed to flow was calculated using a ratio of the free water volume to the total porosity volume of the bed. The results are presented in Table 14. The free water volume usage of the compost media at different flow rates is presented in Figure 14. Neither the compost only nor the vegetated compost beds had a linear relationship between flow rate and volume of bed used. The percentage of treatment bed volume used between flow rates of 70 and 909 mL/min indicates that as flow rate increases so does the amount of free water present in the bed. At the higher flowrates, the volume of free water is higher in the compost only bed than in the vegetated compost bed (Table 14). At the low flow rate of 70 mL/min, the volume of free water was lower in the compost-only bed than in the vegetative compost bed.

Table 14- Percentage of Treatment Bed Utilized at Various Flow Rates (n=3, bed length= 65cm, average value shown)

Flow Rate	Compost Only	Vegetated Compost
Low flow storm event (70mL/min)	5.4%	6.5%
1-year storm event (264mL/min)	14.8%	13.0%
2-year storm event (676 mL/min)	34.9%	21.0%
5-year storm event (900 mL/min)	41.3%	23.8%

*on a slope of 12°

5.2 Sorption Capacity Experiments

The following section presents the results obtained from the experiments designed to quantify the zinc sorption capacity of the two treatment bed configurations. Effluent pH and turbidity are also examined. There was variability in both the moisture content of the treatment beds (due to the nature of stopping and starting the trials) and the influent concentration (see discussion for detailed explanation).

5.2.1 Effect of Vegetation on Zinc Sorption Capacity

The measured zinc concentration of effluent was plotted versus time to compile breakout curves (see Figure 16 and Figure 17 below). Appendix I contains the data sets for both Trial 1 and Trial 2 and Appendix J contains the quality control data corresponding to the zinc sorption trials. During the higher flow rates, 676 and 900 mL/min flow rates, both the vegetated compost and compost only beds experienced a backup of water between the flow distributor and the treatment bed which resulted in pooling. The wetting front of the water on the treatment beds was not measured but varied considerably from trial to trial. The health of the vegetation in the treatment beds declined over the zinc sorption experiment however root network remained intact. The vegetation went from a vibrant green colour to a spotty, yellowish green. The moisture content of the vegetation also declined and the vegetation became brittle and dry towards the end of the trials. After chemical breakout was achieved, the treatment beds were visually examined and a white crystalline powder was present throughout compost after exhaustion in both the compost only and vegetated beds (see Figure 15).

Figure 15- Observed Crystals on Compost

The time for breakthrough was based on total elapsed storm time, disregarding the time between experiments in which no run was being conducted. The total runtime until chemical breakout was approximately 14 880 minutes in length for both the vegetated compost and compost only bed for Trial 1. Additionally, both beds exhibited linear trends in their concentration versus time graph (Figure 16). For practical purposes, the data from both vegetated compost and compost only beds overlapped with no significant difference in chemical breakout time.

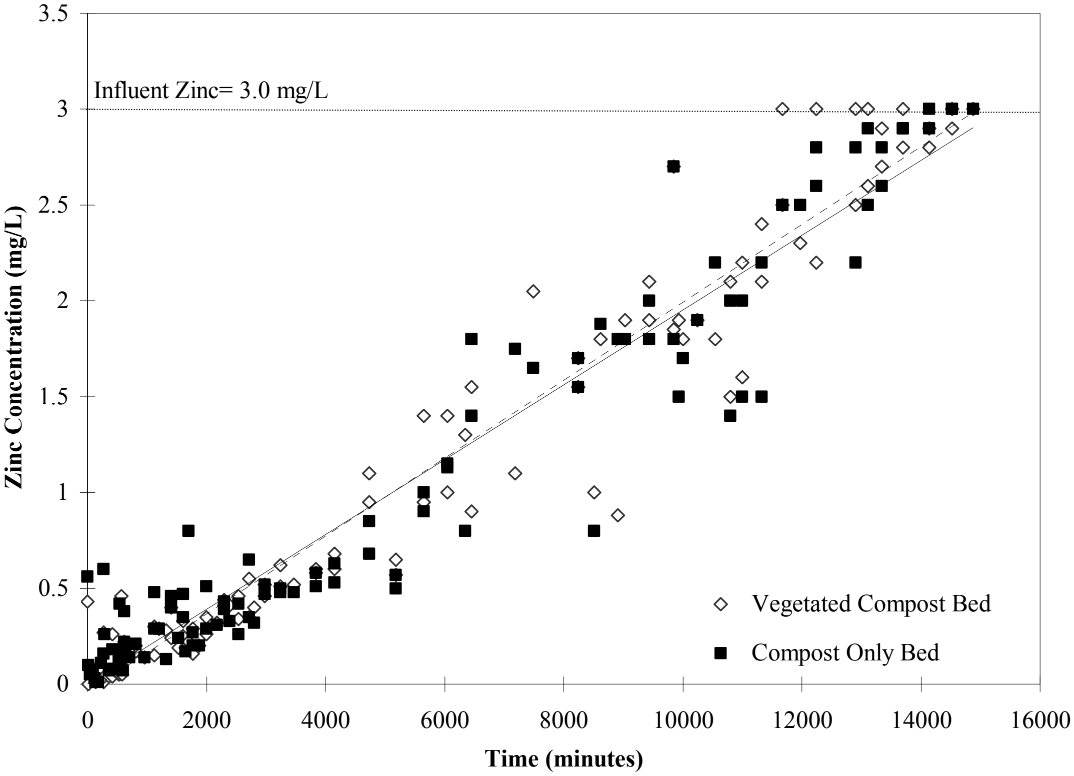


Figure 16- Effluent Zinc Concentrations: Trial 1 (264 mL/min, bed length=65cm)

The time for breakthrough was based on total elapsed storm time, disregarding the time between experiments in which no run was being conducted. The total runtime until chemical breakout was approximately 13 785 minutes in length for both the vegetated compost and compost only bed for Trial 2. Additionally, both beds exhibited linear trends in their concentration versus time graph (Figure 17). For practical purposes, the data from both vegetated compost and compost only beds overlapped with no significant difference in chemical breakout time.

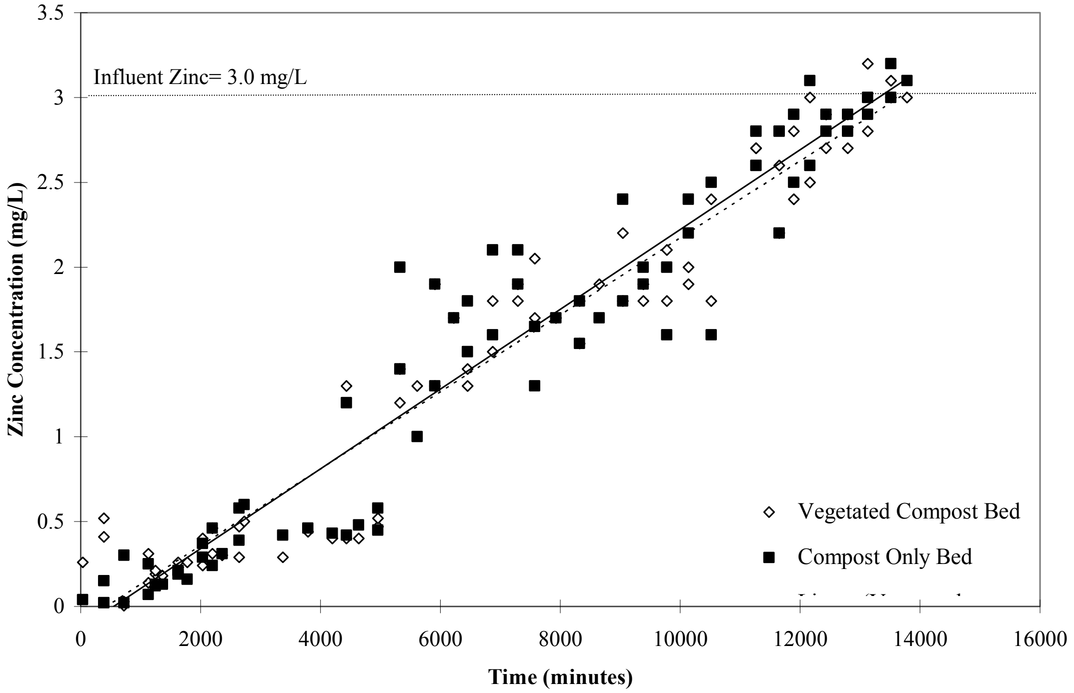


Figure 17- Effluent Zinc Concentrations: Trial 2 (264 mL/min, bed length=65cm)

5.2.2 Effect of Vegetation on Effluent Turbidity

During the zinc sorption experimental trials, the turbidity of the effluent was monitored. Effluent turbidity samples were taken at the start time of each watering event as well as monitored throughout the duration of the event. An initial spike of greater than 20 FTU was observed both compost only and vegetated beds for each watering event and is illustrated in the graphs. The runtime (any turbidity reading taking place after the start of the event) turbidity reading of a watering event the turbidity was significantly lower and less than 20 FTU. It was observed that both the compost-only and vegetated compost beds experienced turbidity decreasing over time. The effluent turbidity at the beginning of the watering events for Trial 1 ranged from 21.6 to 26.4 FTU for initial readings and 16 to 14 FTU for samples taken during a run event. The effluent turbidity for Trial 2 ranged from FTU for initial readings and FTU for samples taken during a run event.

The effluent turbidity readings from Trial 1 are presented in Figure 18 and show a split in data between the high initial readings and subsequent lower runtime readings for. Figure 19 and Figure 20 show the initial and the runtime effluent turbidity readings separated. Both the initial and runtime turbidity data from Trial 1 appear to exhibit an asymptotic curve.

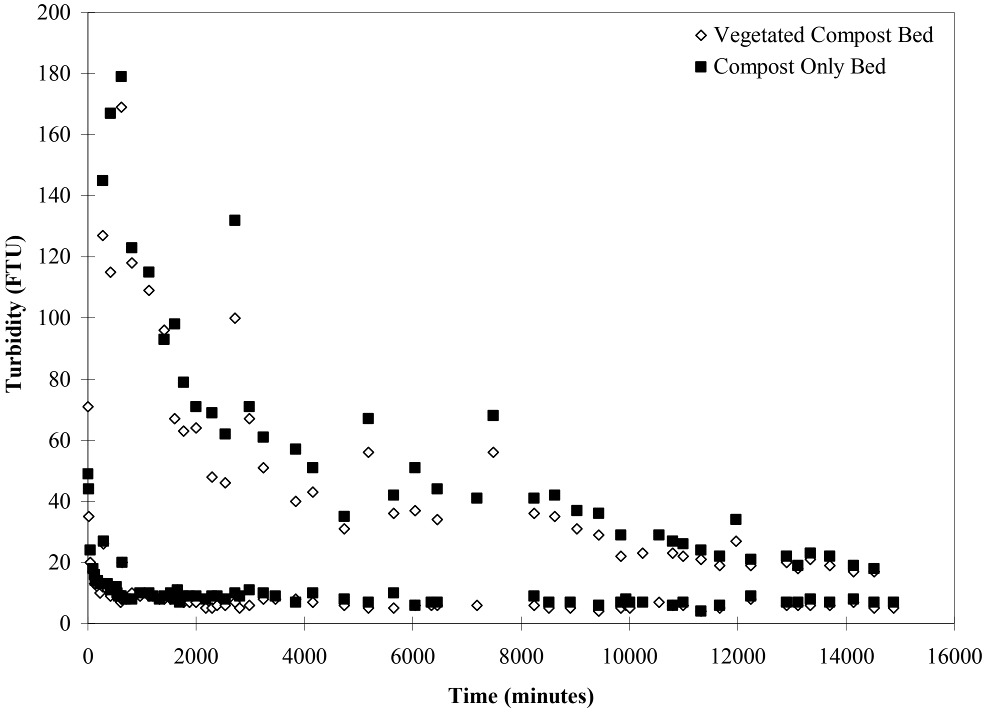


Figure 18- Effluent Turbidity: Trial 1 (264 mL/min, bed length=30cm)

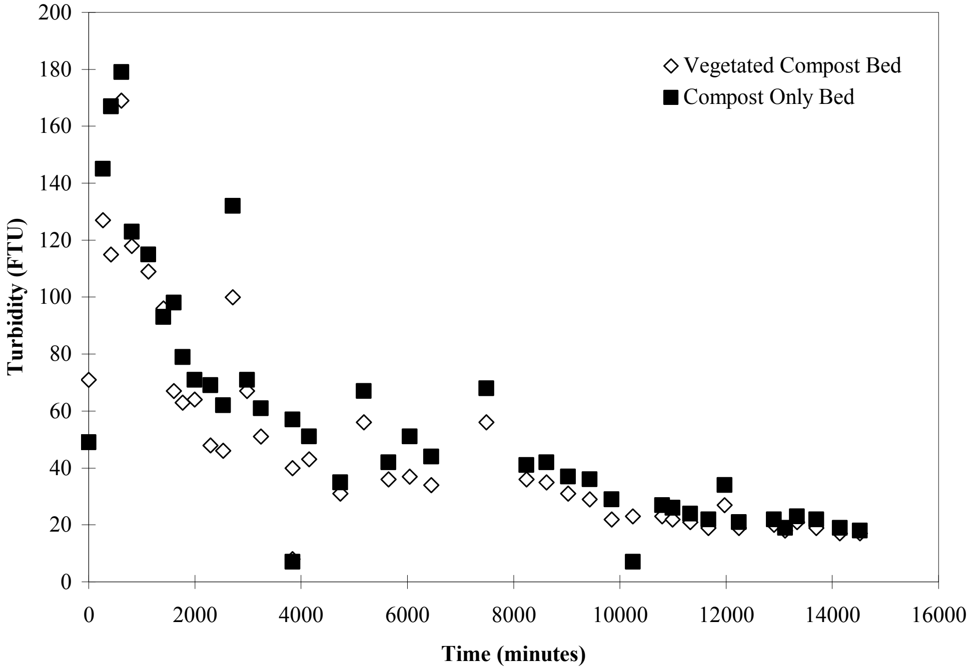


Figure 19- Initial Effluent Turbidity: Trial 1 (264 mL/min, bed length=30cm)

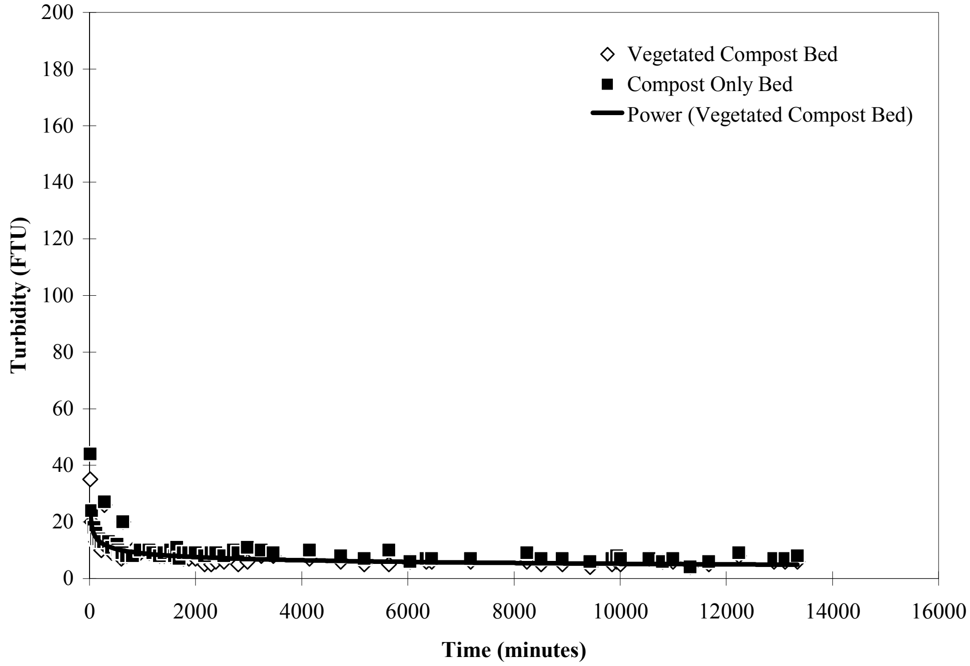


Figure 20- Runtime Effluent Turbidity: Trial 1 (264 mL/min, bed length=30cm)

The effluent turbidity readings from Trial 2 are presented in Figure 21 and show a split in data between the high initial readings and subsequent lower runtime readings for. Figure 23 and Figure 22 show the initial and the runtime effluent turbidity readings separated. The initial turbidity data from Trial 2 appears to decrease before leveling off whereas the runtime turbidity data appears to remain relatively constant over the course of the trial.

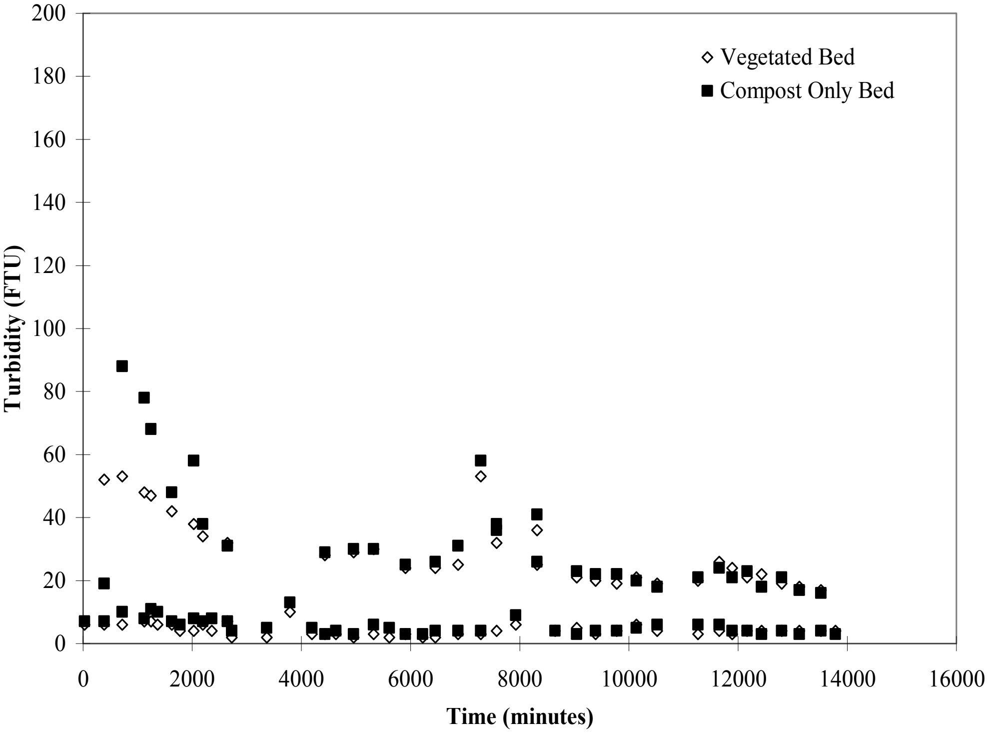


Figure 21- Effluent Turbidity: Trial 2 (264 mL/min, bed length=30cm)

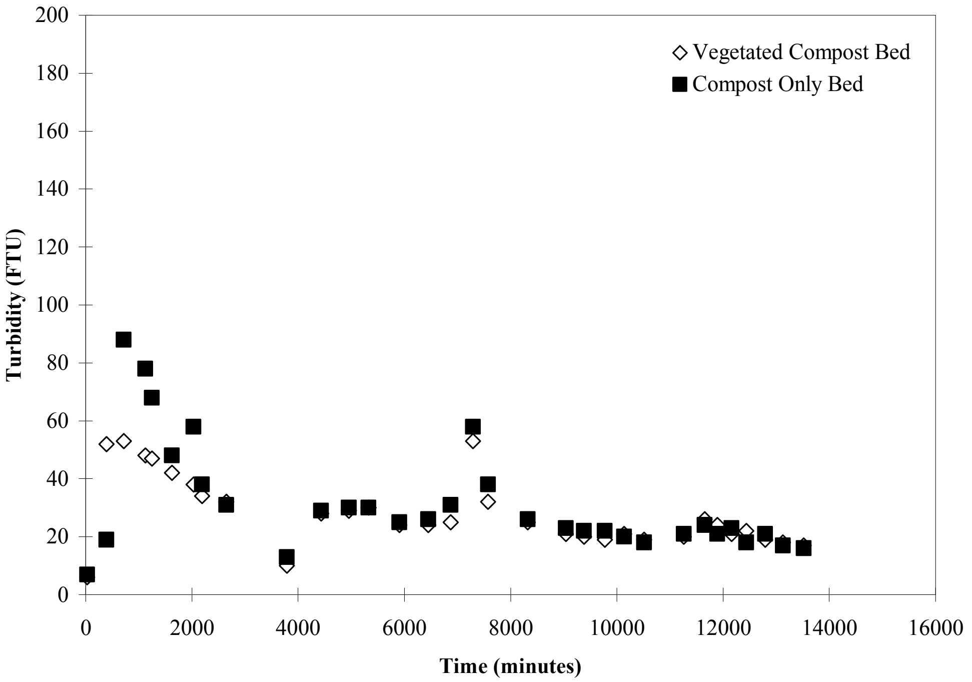


Figure 22- Initial Effluent Turbidity- Trial 2 (264 mL/min, bed length=30cm)

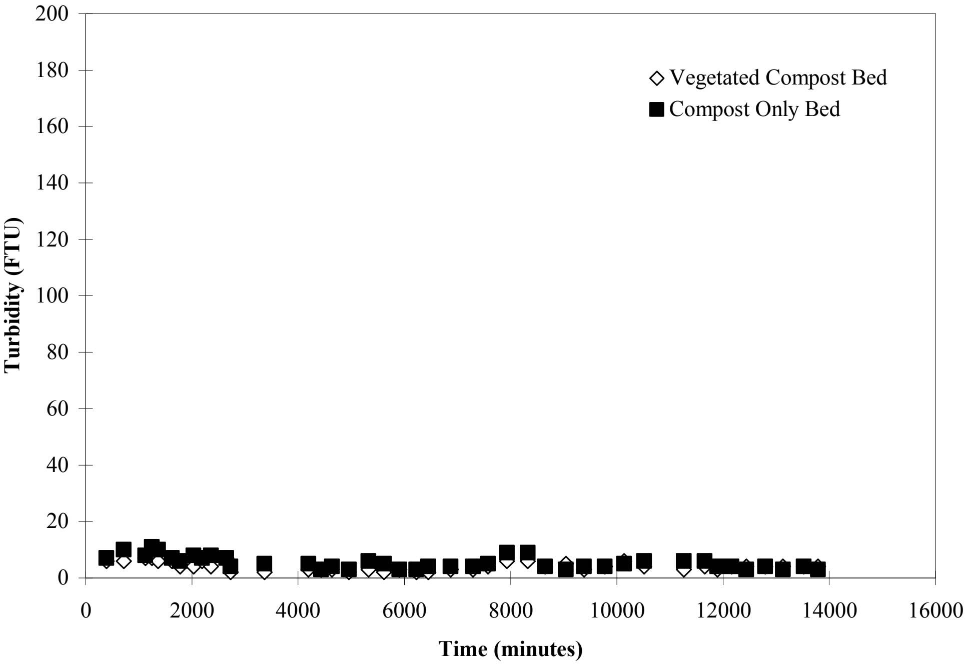


Figure 23- Runtime Effluent Turbidity: Trial 2 (264 mL/min, bed length=30cm)

In order to determine whether the observed differences in turbidity between vegetated beds and compost only beds were significantly different, data were subjected to two tailed t-test, using MS Excel. Data from Trial 1 was subjected to a paired t-test between the initial effluent turbidity readings of vegetated compost and compost only beds which indicate that the critical value of t (3.54) obtained from t distribution table at 1% probability level (df=41) is lower than observed value of t-stat (4.37). Therefore the differences between the pairs of initial startup turbidity readings were significant and the vegetation reduced turbidity by 13.9 % on the average. Almost similar results were found for the initial effluent turbidity readings of Trial 2. A two-tailed t-test utilizing all of the data from Trial 2 initially shows that the differences between the pairs of turbidity readings and zero were not significant for initial readings (tstat=1.70, p=0.099, df=31). However, after examining the graphical data, the differences between the pairs of initial startup turbidity readings and zero were significant if the first two readings and the last ten were not included in the data set (tstat=3.18, p=0.005, df=19). The vegetation reduced turbidity by 18.9% on the average. Thus, it can be concluded that real and significant differences exist between the vegetated compost bed and the compost only bed for initial turbidity readings of effluent samples.

Two tailed t-tests also conducted for midrun turbidity readings indicated that the differences between the pairs of midrun turbidity readings for Trial 1 were significant (tstat=8.6, p<<<0.0001, df=68) with vegetation reduced turbidity by 15.6 % on the average. Similarly, the differences between the pairs of midrun turbidity readings for Trial 2 were significant (tstat=5.44, p<<.0001, df=41) with vegetation reducing turbidity by 30.4 % on the average. Additionally, there was a significant difference found between the regression slopes of the compost only and vegetated compost bed in terms of effluent turbidity with vegetation being lower. Thus, it can be concluded that real and significant differences exist between the vegetated compost bed and the compost only bed for midrun turbidity readings of effluent samples as well. Detailed statistical tables for all analysis are presented in Appendix K.

Further analysis was conducted on 4 randomly chosen effluent samples in order to determine the contribution of colour to the overall turbidity. Colour is defined as the organic material that has dissolved into solution, while turbidity consists of tiny particles suspended in the water column. Turbidity has the capacity to settle out over time whereas colour remains uniform throughout liquid. Total turbidity and dissolved turbidity were analyzed before and after filtering through 0.45 μm acid-washed filters using both the colorimeter (Formazine Turbidity Unit (FTU)) and turbidity meter (measuring in Nephelometric Turbidity Unit (NTU)). Results below in Table 15:

Table 15- Dissolved Turbidity and Total Turbidity

	Unfiltered= Total Turbidity		Filtered= Dissolved Turbidity=Color	
Sample	Colorimeter (FTU)	Turbidity Meter (NTU)	Colorimeter (FTU)	Turbidity Meter (NTU)
Initial Sample Compost Only	38.00	13.00	28.00	10.00
	37.00	12.50	27.00	10.00
	37.00	13.00	27.00	10.00
Initial Sample Vegetated Compost	34.00	8.80	26.00	5.10
	35.00	8.40	26.00	5.30
	34.00	8.70	26.00	5.40
Runtime Sample Compost Only	3.00	0.40	2.00	0.40
	3.00	0.40	2.00	0.30
	3.00	0.45	2.00	0.35
Runtime Sample Vegetated Compost	2.00	0.45	3.00	0.50
	3.00	0.45	2.00	0.30
	3.00	0.50	3.00	0.20

The results of the colorimeter analysis indicated that 73.60 (+/-18.7, n=3, 1 standard deviation) % of the total turbidity could be attributed to dissolved colour and not suspended solids. The results of the turbidity meter analysis were similar showing 80.29(+/- 23.7, n=3, 1 standard deviation) % of the total turbidity was attributed to colour and suspended solids. In conclusion, colour is the main contributor to total turbidity.

5.2.3 Effect of Vegetation on Effluent pH

During the zinc sorption experimental trials, the pH of the effluent was recorded over time. It was observed that both the compost-only and vegetated beds had a buffering effect on the metal influent stream. The incoming stream was kept constant pH of 5.6 and the effluent pH was initially in the range of 6.5-7.0 before gradually decreasing as the treatment beds approached exhaustion. The effluent pH results from Trial 1 are presented in Figure 24. The compost only treatment configuration buffered the pH of the influent to 6.46 (+/- 0.37, n=111, 1 standard deviation) and the vegetated compost treatment configuration buffered the pH of the influent to 6.39 (+/- 0.46, n=111, 1 standard deviation). The pH of the effluent of both beds declined slightly over the lifetime of the treatment beds to a final pH of 5.85 for the compost bed and 5.89 for the vegetated compost bed.

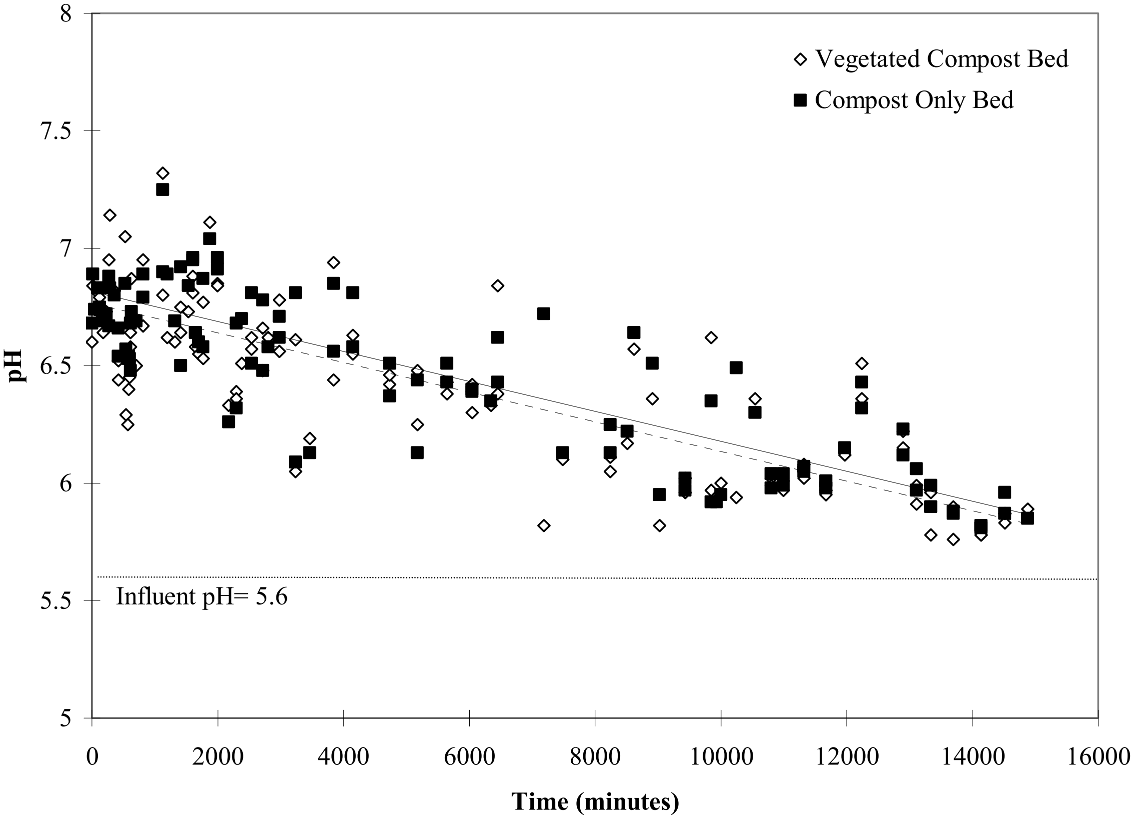


Figure 24- Effluent pH: Trial 1

The effluent results from Trial 2 are presented in Figure 25. The compost only configuration buffered the pH of the influent to 6.41 (+/- 0.23, n=76, 1 standard deviation) and the vegetated compost treatment configuration to 6.40 (+/- 0.22, n=78, 1 standard deviation). The pH of the effluent of both beds declined slightly over the lifetime of the treatment beds to a final pH of 5.96 for the compost bed and 5.94 for the vegetated compost bed.

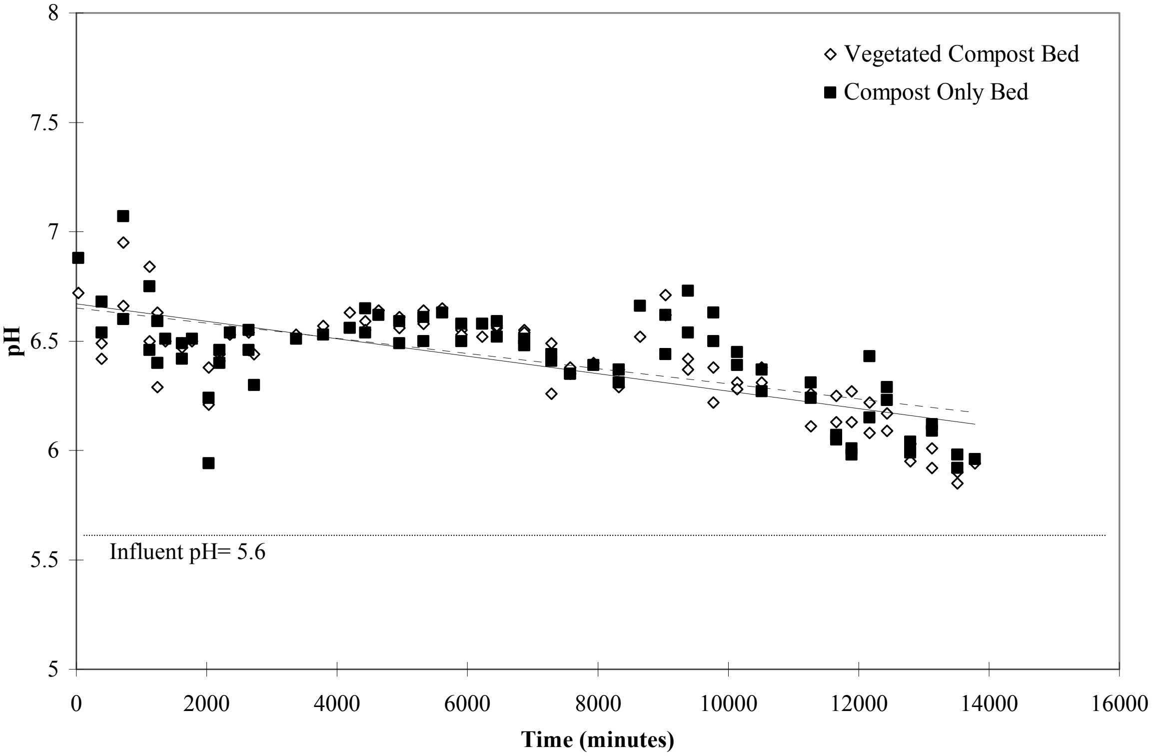


Figure 25- Effluent pH: Trial 2

In order to determine whether the observed differences in pH over time were significant in the vegetated beds and compost only beds an analysis of variance (ANOVA) was performed using MS Excel. Regression analysis was preformed on data from Trial 1 which indicated a positive relationship between change in pH of the vegetated compost bed effluent and time (F=164, p<<0.0005, df=1, 109). The same analysis was carried out for the compost only bed which also indicated a positive relationship between effluent pH and time (F=200, p<<0.0005, df=1, 109). Finally, the regression slopes were compared between the compost only and vegetated compost beds.

Similar results were found for the pH turbidity readings of Trial 2 There is a positive relationship between change in pH of the vegetated compost bed effluent and time (F=88.21, p<<0.0005, df=1,72). There is also a positive relationship between change in pH of the compost only bed effluent and time (F=48.5, p<<0.0005, df=1,72). Additionally, there was no significant difference found between the regression slopes of the compost only and vegetated compost bed in terms of effluent pH. Thus, it can be concluded both the vegetated compost bed and the compost only pH effluent samples have a significant relationship with time. Detailed statistical tables for all analysis are presented in Appendix K.

6.0 Discussion and Potential Applications

This chapter is used to discuss the results obtained from the experiments conducted. Similar previously conducted research is compared and contrasted when appropriate, and novelty of the results is highlighted. Based on the result analysis, data from replicate trials were different due to variability in soil moisture content, initial wetting front height and time between trial runs. As these variations would likely be evident in a real situation, therefore the variations in experimental conditions were permitted.

The "S"-shaped C(t) curves generated by the bromide tracer experiments were found to be similar to what is expected in theory (Figure 26), similar to non-ideal plug flow reactor (PFR) with dispersion in y direction (observed with wetting front). Pooling was observed in both treatment beds at flow rates greater than 70 mL/min and may be explained by the need for water to build up potential energy in order to overcome the resistance of the compost media (see images in Appendix L). In open channel hydrology, water rises in a channel to gain enough energy to overcome its kinetic losses which is observed in both treatment beds. The RTD curves do not show heavy tailing which would be indicative of stagnant water pooling. The "S" curve is preserved and the step change is completed ($C_{in}=C_{out}$) in all trials. The "S" shape curve of the C(t) graphs do not exhibit short-circuiting which would be indicative of channeling.

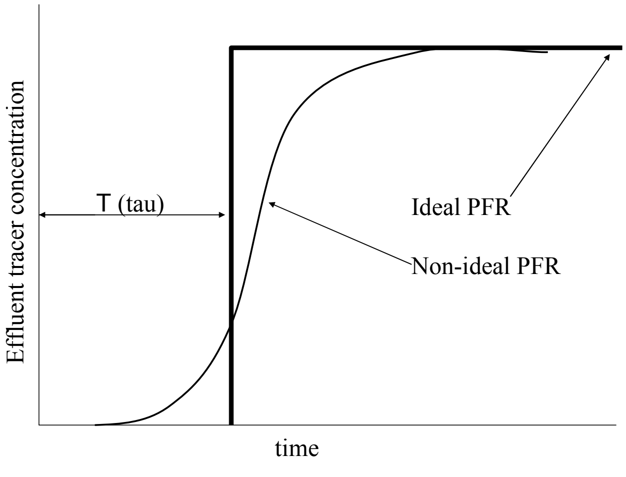


Figure 26- C(t) curves of ideal versus non-ideal flow in a PFR

Results from the RTD graphs generated specific retention times from the flow characterization experiments and indicated that retention time is only effected at the low flow rate of 70 mL/min when the retention time for the vegetated compost bed was greater than that of the compost bed only bed. This may be attributed to the relationship between the drag force provided by the root structure present in the vegetation and velocity. At lower velocity (flow rate 70 mL/min) root mass present at the bottom of the bed offered higher resistance to flow (Figure 8). However as velocity of flow increases, water in the vegetated bed would have increased beyond that mass of roots (vertically). At that time, flow resistance in both beds would have been similar and velocity would have been similar and velocity would have likely been the main contributing factor due to the V^2 term in the general drag formula (Equation 7).This is consistent with the RTD graphs and retention times of the higher flow rates being not significantly between bed configurations.

$$F_D = \frac{1}{2} C_D A \rho V^2 \qquad \text{(Alexandrou 2001)} \qquad [7]$$

F_D =drag force (Newton)

C_D =drag coefficient (dimensionless)

A =cross sectional area perpendicular to the flow

ρ =density of the liquid medium (kg/m^3)

V =velocity of the body relative to the medium (m/s)

Tortuosity may also be a contributing factor to the difference in retention times. Tortuosity is a property of curve being tortuous (twisted; having many turns) and is commonly used to describe diffusion in porous media (Epstein 1989). At low velocities, the bulk of the flow is flowing along bottom of the bed where the root system is occupying a portion of the cross sectional flow path. The resistance created by the twists and turns of root system in the vegetated bed maybe contributing to the tortuosity of the treatment bed.

The free water volumes of the treatment beds shown in Figure 14 agree with the findings of the tracer study and are supported with theory. The vegetated compost contains less pore space because of the existing root structure. At the low flow rate, less pore space means more resistance causing the water to move in the y-direction (Figure 27 and Figure 28) and as a result, the vegetated compost has a longer retention time and higher free water volume. The root mat at the bottom of the vegetated bed also adds resistance to flow at the low flow rate (Figure 8).

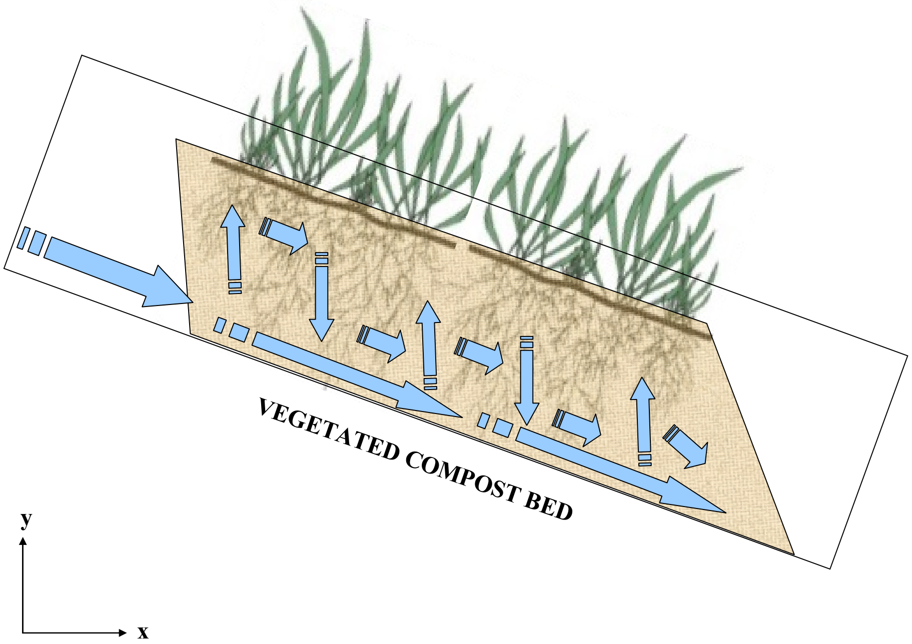


Figure 27- Vegetated Compost Bed, 70mL/min flow, roots adding resistance to flow

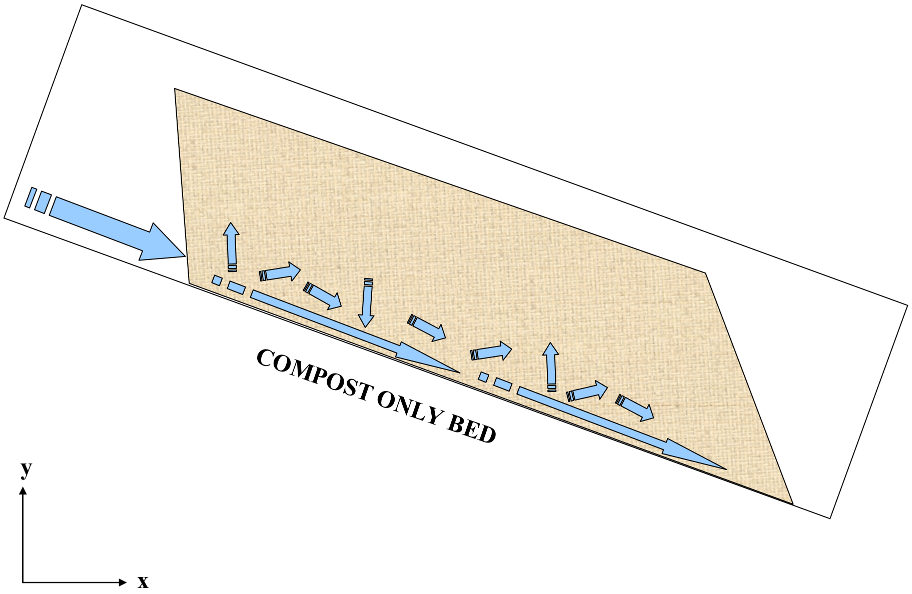


Figure 28- Compost Only Bed, 70 mL/min, less resistance to flow

At the flow rates greater than 70 mL/min, the compost-only bed has a higher free water volume which may be attributed to the higher porosity due to the absence of root structures. In the vegetated compost bed the higher flow rates are at a velocity that can overcome the resistance of the root structure which results in similar retention times to the compost-only configuration (Figure 29 and Figure 30).

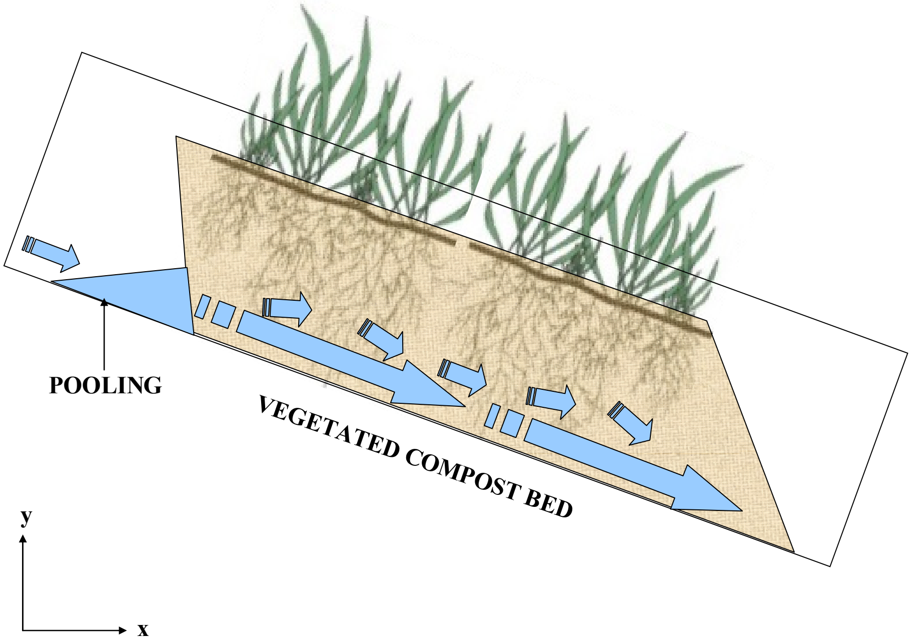


Figure 29- Vegetated Compost Bed, >70mL/min flow, root resistance overcome by higher velocity of flow

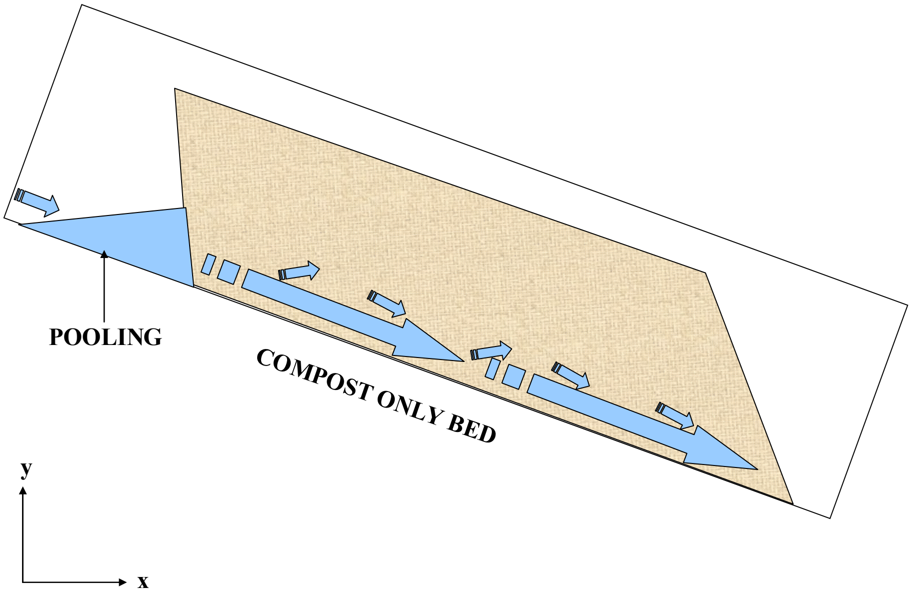


Figure 30- Compost Only Bed, >70 mL/min

The results from section 5.1.2 compared the retention times of treatment beds at different lengths. The retention times did not relate linearly to bed length. Previous sorption column experiments and theory (Geankoplis 1993), has shown that porous media in columns can be scaled by length and is a standard recommendation for columns (both upward and downward flow). These experiments did not take into account potential capillary action which may account for the differences between being able to scale a column by length but not be able to scale an open treatment bed. The constant height of the mass-transfer zone can be used in scale-up when the height of the overall bed is large relative to the mass transfer zone (Geankoplis 1993). In the application outlined here, the bed is sloped, and not all of the bed is used, so that standard recommendation cannot be used. More work is needed to understand scaling in this application. Using the information available currently, treatment beds should be scaled conservatively.

Zinc sorption experiments were conducted at a flow rate of 264 mL/min showed considerable overlap in both trials (Figure 16 and Figure 17) and the results yielded no significant difference in chemical breakout time between the two treatment bed configurations. At this flow rate, the residence time distribution curves overlapped for the compost only and vegetated compost treatment beds. The retention times (87.92 and 93.61 seconds) and pore volume usage (14.8% and 13.0%) were very similar at this flow rate. The grasses in the vegetated compost bed acted like metal tolerant species which tend to behave like metal excluders instead of accumulators (Bake 1981). No literature was found to support ion exchange of metals occurring in dormant vegetation so it is likely no metals were removed by the vegetation. It would be beneficial to explore zinc sorption experiments at the lower flow rate of 70 mL/min where large differences were found in retention time and pore volume usage. Moreover, in the context of field application the majority of rain events in the GVRD take place at intensity less than 264 mL/min, (which is a 1 year storm event). As previously mentioned in section 4.2.2, the incoming zinc concentration did have some variation (Appendix J). This accounts for some of the spikes and dips seen in the effluent graphs (Figure 16 and Figure 17. which would be a reflection of real-life conditions in the field. It is worth noting that the zinc sorption profile created reflects compost that was kept fairly moist due to the successive watering application. During field experimentations, the compost would be expected to dry out between rainfall events. It would be beneficial to evaluate this in future work. It should also be noted that although the zinc concentration and flow rate selected for the runoff simulations correspond to literature runoff values, these values occur during the first flush of the storm event and not the duration. The first flush of the storm is the initial period of the storm event that carries the highest pollutant concentration and lasts approximately 20 minutes (Lee et al. 2002; Gupta and Saul 1996). Thus, the conditions used in this research are equivalent to multiple first flushes occurring in succession therefore emulating an extreme field scenario.

Zinc was chosen in order to evaluate the worst case scenario as the most mobile metal ion. If a less mobile ion (nickel or aluminum) was used, the results may have been different. The interference of the flow caused by the vegetated roots may have ‘trapped’ the less mobile metals better than the most mobile metal, zinc. Therefore, if the sorption capacity experiments were repeated with a low level mobility metal, the compost-only bed may have reached chemical breakout first.

One of the limitations of this zinc sorption experiment is that only one metal was used and it does not account for the potential impact of the competitive adsorption on that is possible with multi metal stormwater runoff. Additional research into multi-metal trials would be beneficial. Other factors that were not address in this experiment include;

- changes in flow rates during storm events,
- changes in flow characteristics as the site “ages”,
- the effect of precipitation infiltration, and
- vegetative uptake of metals.

The health of vegetation declined during the trials, which was to be expected under the conditions of low light and lack of nutrients during experimentation however vegetation was not examined after trials had ceased to determine if recovery was possible. The deterioration of vegetated health was most likely attributed to lack of adequate light and fertilizer and not necessarily the exposure of zinc. The constant application of water over the trials may have also contributed to the decline of vegetative health. Prior to the start of the experimental trials, the vegetation was green and appeared healthy. After the trials, the vegetation appeared yellow and dry. This indicated that the vegetation was most likely in a dormant and non-metabolically active state and therefore vegetation did not uptake metal either passively or actively (see Appendix C for potential metal uptake mechanisms used by plants).

Effluent turbidity during sorption experiments was found to be lower in the vegetated compost treatment beds than the compost only treatment beds. This may be explained by root structure in the vegetated beds binds and retains small particles that contribute to turbidity. During the watering event, it was observed that the turbidity decreased from the initial peak after the compost particles became wet. Initial effluent turbidity was observed to be higher when bed is left for longer than 24 hours (compost dries out) for both compost-only and vegetated compost beds It is possible that after this wetting, the thin film of water on the compost particles was providing a resistance to mass transfer between the water descending through the bed and the wet compost particles. Moreover, when water comes in contact with dry compost, the humic compounds present in the compost leach out and contribute colour to the effluent in the form of organic humic acids. Another trend identified was that over time, as the beds exhausted, both the initial turbidity readings and subsequent turbidity readings from watering events decreased. This phenomenon may be able to be used as an indicator to identify when the sorption capacity of treatment beds is exhausted

The pH of the influent was buffered by both the vegetated compost and compost only beds. Over time, the pH of the effluent decreased slightly. The decrease in pH is most likely attributed to the ion exchange capacity of the media reaching an end. When the bed approaches exhaustion, the ion exchange of zinc for calcium and potassium ceases and zinc is left in solution. As less zinc ions are removed from the influent stream by ion exchange and sorption, more ions are present in the effluent stream, leaving the effluent slightly more acidic. This phenomenon may have the potential to be used as an indicator of chemical bed exhaustion.

A traditional turbidity meter not available during the trials therefore the Smart 2 Colorimeter was used to take relative turbidity samples. It is recommendation for future work to have samples analyzed with turbidity meter. Also, turbidity should be monitored aggressively from initial water contact until turbidity lowers and stabilizes to runtime turbidity in order to observe the complete turbidity time profile.

Despite having similar hydrodynamic and sorption properties, the addition of vegetation is considered advantageous because it is capable of combating erosion and may increase the lifetime of the system by minimizing compost loss to high flow precipitation events. The presence of vegetation in roadside compost strips allows higher flow rate without fluidization and offers comparable metal sorption rates and pH buffering capacities while providing a lower effluent turbidity. Additionally, because the vegetated compost configuration has significantly less pooling behind the treatment bed, it has the potential to lower the possibility of road flooding at high storm intensities.

The white crystalline substance observed on both the vegetated compost and compost only beds after chemical breakout (Figure 15) could be attributed to zinc precipitating out of solution and accumulating over time. (Spark et al. 1997) found that at a pH of 8.4, 50% of 0.0001 M zinc solution precipitates out. In the 5-7 pH range found in these experiments, 4-5% precipitates out of solution and the peak sorption for humic acids 20-40% of 0.001 M, bound as insoluable zinc hydroxides (Spark et al. 1997). The pH of the runoff plays an important part in the sorption of zinc because the hydrogen ions in a highly acidic solution will compete for the binding sites in the compost bed and may even detach the already bonded zinc ions. The adsorption of metals is highly pH dependent and the dependence is nonlinear. It increases from near zero to near 100% saturation over a relatively narrow critical pH range of 1-2 pH units. This means that a small change in pH can cause a sharp increase in dissolved metal levels. The pH depends, to a large degree, on the buffering capacity of the system. If they buffering capacity is slowly consumed, as may happen as a result of a continuous input of acidic atmospheric precipitation or due to acidic stormwater runoff, a certain threshold will be exceeded and metal will begin to suddenly desorb from the sediments or the soils. The crystalline phenomenon requires further investigation.

The results found in this report cannot confidently recommend vegetated compost as a roadside stormwater BMP without additional work because the metal sorption was examined for zinc only and not the full spectrum of metals present in stormwater runoff. Additionally the large scale application necessary for roadside treatment does not allow for monitoring of longevity. Moreover, more research needs to conducted as site ‘ages’ with plant growth and on determining the end product of chemically exhausted beds is stable as well as appropriate disposal options although disposal options may not be necessary. According to (Walsh et al. 2007), the rate of zinc and lead deposition from highway runoff on the filter strips is less than one-tenth the maximum deposition rate allowed by the 503 Regulations, which limit application rates of metals in biosolids to cropland. Any threats to human health and to the environment from metals deposition from highway runoff on vegetated areas are small. Accumulation of metals in the monitored filter strips could continue for over 200 years without risk.

Potential applications that can be recommended based on the results of this research include industrial runoff, galvanized roofs, galvanized yard equipment on concrete pad or low concentration mining when zinc as metal of concern and the concentration and flow rates are known and similar to the conditions examined in this report. The cost of installing a vegetated compost system is approximately \$19-21 a cubic meter for compost delivered to site and \$4-6 meter squared for blower truck application of compost and seed have been completed. The vegetated compost treatment beds should be scaled based on width and use a conservative length of 65 cm. At an application rate of 264 ml/min at a concentration of 3.0 mg/L of zinc, the effluent of the system would reach compliance in approximately 2000 minutes (continuous runtime). Exhausted beds could be disposed of in a waste to energy facility until end use studies. Site specific design criteria would have to be gathered and processed in order to more accurately predict the performance of the system. A conceptual diagram can be seen in Figure 31. The addition of a gravel precursor is shown which may be beneficial as a primary screen for large objects as well as assisting even flow distribution. The gravel would also act as a reservoir in the case of a high flow storm that may cause pooling behind the vegetated compost bed.

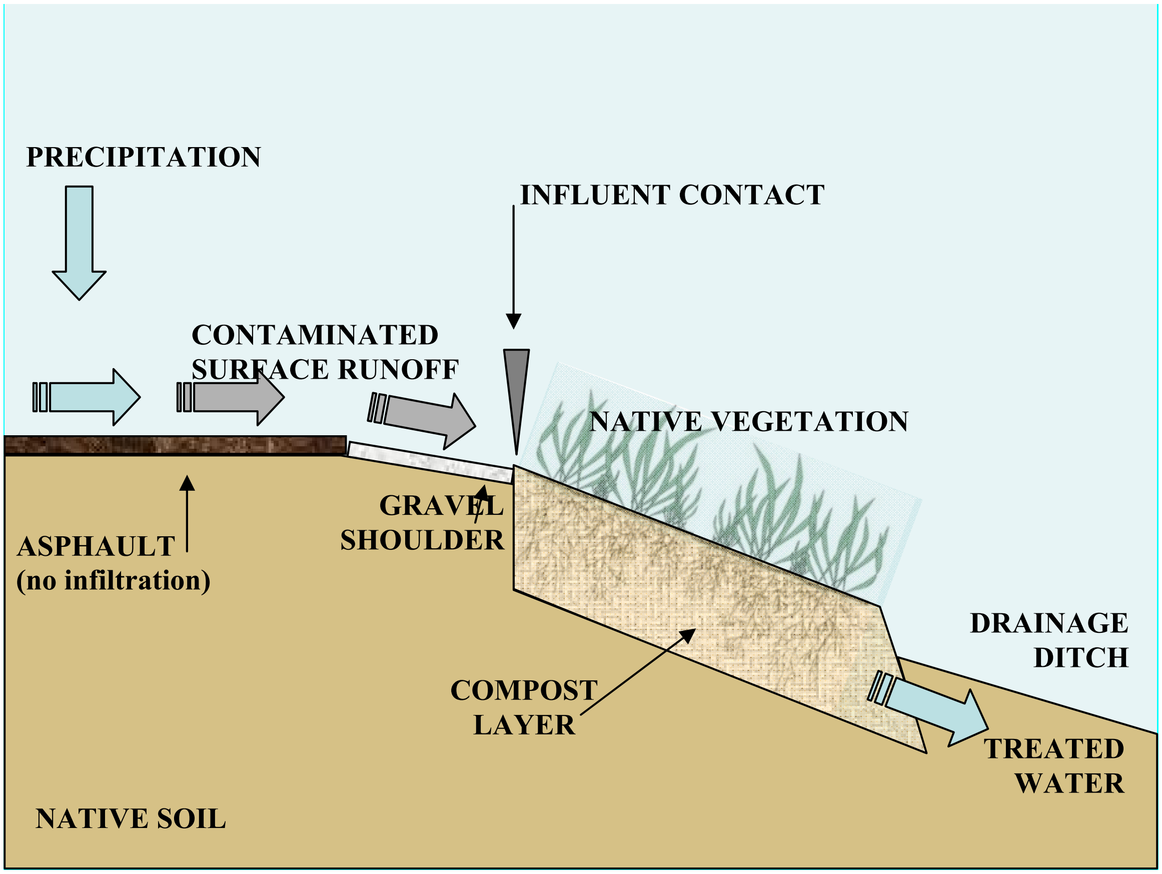


Figure 31- Conceptual Drawing of Contaminated Runoff through Vegetated Compost

Based on the results presented, highway grass strips can act as an effective media sorbant for metals and also produce a lower turbidity effluent compared to a non-vegetated strip (when yard waste compost is used). The use of vegetated compost strips could be utilized now for to lower turbidiy, increase retention time (in runoff events less than 70 ml/min) and buffering the pH of influent stormwater in the GVRD. It should be noted that the actual longevity capability of a field site application would be significantly longer than laboratory treatment beds because experimental trials were conducted at a constant flow rate of 264 mL/min, the equivalent to a storm intensity of a 1 year storm event whereas actual precipitation patterns are sporadic with varying intensity. It can be hypothesized that significant metal retention would be anticipated for times significantly greater than the experimental period. As with all stormwater controls, a careful evaluation is needed for determining if a method or device should be used for a specific site application. Consideration must be given to structural and media hydraulics, media type, site limitations, structural constraints and the need for future maintenance.

Further study into the effects of organic matter, plant speciation, ground cover, and moisture conditions on metal retention would also greatly enhance the understanding of metal migration and ultimately contribute to the design of roadside vegetated strips for use at their highest potential. It is recommended, based on the positive results obtained herein, that a long-term field study be initiated to extend the data base to multi-metal conditions and a longer flow history. This would afford a means of evaluating variability that can only be experienced in the field and result in a confirmation of the laboratory studies, lending support to design criteria.

7.0 Conclusions and Recommendations for Future Work

The tracer tests were used to successfully determine retention times for laboratory-scale roadside stormwater filter strips containing compost only and dormant/ non-growing vegetated compost at flow rates between 70 and 900 mL/min. Additionally, zinc sorption experiments gave insight into the effluent concentration profile, pH, turbidity and longevity of the treatment beds. This information can be used to design new BMPs within the temperate rain forest region that extends from central coastal California to South Alaska. Findings from the experiments are summarized below.

Flow characterization of roadside filter strips

Flow characterization experiments investigated stormwater runoff rates of 70, 264, 676 and 900 mL/min flow rates, in order to evaluate the effect of vegetation on subsurface flow conditions. Results from flow characterization experiments that were used to evaluate the effect of roots on subsurface flow in sloped beds (treatment media dimensions: L=0.64 m, W=0.254 m, H=0.10 m) with treatment indicated that roots effected retention time only at the low flow rate of 70 mL/min. Then, the retention time for the compost bed was less than that of the vegetated compost bed. The hydraulic retention times calculated ranged from a high of 417 s for the vegetated compost configuration at a flow rate of 70 mL/min to a low of 59 s at a flow of 900 mL/min for the compost only bed.

Roadside filter strips as sorption material for dissolved metal

The most important result found from this investigation was that there was not a significant difference in the breakout time between the compost only and vegetated compost treatment beds in terms of chemical breakout time when the zinc concentration was 3.0 mg/L and the pH and temperature of the influent was 5.6 and 11°C, respectively. Other parameters investigated during the sorption capacity trials were the turbidity and pH of the effluent. The results indicated that both the compost only and vegetated compost treatment configurations were capable of buffering the pH of the influent to 6.46 +/- 0.37 and 6.39 +/- 0.46 in Trial 1 and 6.41 +/- 0.23 and 6.40 +/- 0.22 in Trial 2, respectively. Additionally, the pH of the effluent of both beds declined slightly over the lifetime of the treatment beds. Therefore it may be concluded that both the compost and the vegetated compost treatment bed are capable of effectively buffering the incoming acidic stormwater runoff. Initial turbidity readings for both treatment beds were found to be significantly higher than the runtime turbidity. Results indicated that the vegetated compost treatment bed produced lower initial and runtime effluent turbidity. Additional analysis concluded that a large portion of the effluent turbidity could be attributed to colour and not suspended solids. A white crystalline powder was observed in the beds at the end of the zinc sorption experiments and the health of the vegetation also deteriorated. Both of these phenomena should be examined further.

Overall, this research project has shown that dormant mature grass vegetation grown on compost produces similar effluent turbidity, pH and zinc sorption profile as a compost only system. Several application recommendations can be made based on the results of this project including industrial runoff, galvanized roofs, galvanized yard equipment on concrete pad or low concentration mining when zinc as metal of concern and the concentration and flow rates are known and similar to the conditions examined in this report.

Information on retention times at low flow rate is quite valuable because it gives insight into the most common storm events in the GVRD which is a flow event with intensity less than 1 year storm return period. The completion of the suggested future work outlined in this report would move the use of the combination of low cost yard waste compost and native dormant mature vegetative grass cover to be a viable BMP to treat metals in highway stormwater runoff.

Suggested Future Work

The following recommendations for future research would give greater depth into the hydrodynamics and sorption components of the treatment configurations outlined in this report:

- Conduct zinc sorption experiments at low flow rates (70 mL/min)
- Determine sorption isotherms for Filterexx Grow Media compost
- Determine chemical composition of white crystalline powder that is present on exhausted treatment beds
- Determine kinetic uptake rates of dormant plants with respect to metals to determine if metal sequestration takes place under dormant winter conditions
- Determine kinetic uptake rates of compost with respect to metals to determine how this rate compares to the uptake rate of vegetation
- Explore the fate of vegetated compost strips over time with respect to metal retention under the conditions of further organic breakdown as the site ‘ages’
- Investigate any potential disposal issues of the chemically exhausted treatment beds
- Investigate if plants are viable after being subjected to metal concentrations while in a dormant state
- Assess the health of vegetation when exposed to multiple metals at concentrations similar to actual stormwater runoff

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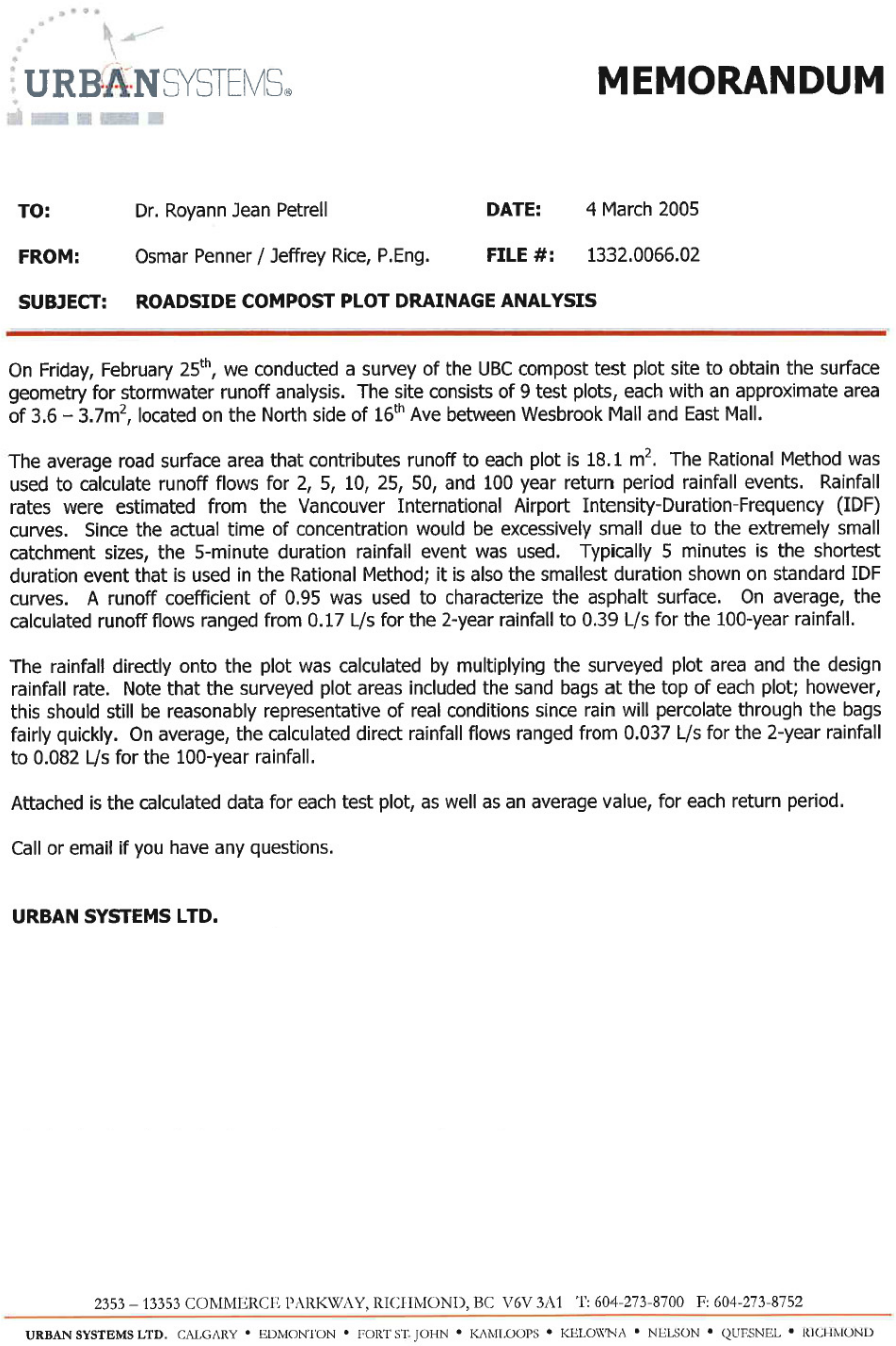
APPENDIX A –Hydrology Data and Precipitation Data for GVRD, British Columbia



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MEMORANDUM

TO:	Dr. Royann Jean Petrell	**DATE:**	4 March 2005
FROM:	Osmar Penner / Jeffrey Rice, P.Eng.	**FILE #:**	1332.0066.02

SUBJECT: **ROADSIDE COMPOST PLOT DRAINAGE ANALYSIS**

On Friday, February 25th, we conducted a survey of the UBC compost test plot site to obtain the surface geometry for stormwater runoff analysis. The site consists of 9 test plots, each with an approximate area of 3.6 – 3.7m^2, located on the North side of 16th Ave between Wesbrook Mall and East Mall.

The average road surface area that contributes runoff to each plot is 18.1 m^2. The Rational Method was used to calculate runoff flows for 2, 5, 10, 25, 50, and 100 year return period rainfall events. Rainfall rates were estimated from the Vancouver International Airport Intensity-Duration-Frequency (IDF) curves. Since the actual time of concentration would be excessively small due to the extremely small catchment sizes, the 5-minute duration rainfall event was used. Typically 5 minutes is the shortest duration event that is used in the Rational Method; it is also the smallest duration shown on standard IDF curves. A runoff coefficient of 0.95 was used to characterize the asphalt surface. On average, the calculated runoff flows ranged from 0.17 L/s for the 2-year rainfall to 0.39 L/s for the 100-year rainfall.

The rainfall directly onto the plot was calculated by multiplying the surveyed plot area and the design rainfall rate. Note that the surveyed plot areas included the sand bags at the top of each plot; however, this should still be reasonably representative of real conditions since rain will percolate through the bags fairly quickly. On average, the calculated direct rainfall flows ranged from 0.037 L/s for the 2-year rainfall to 0.082 L/s for the 100-year rainfall.

Attached is the calculated data for each test plot, as well as an average value, for each return period.

Call or email if you have any questions.

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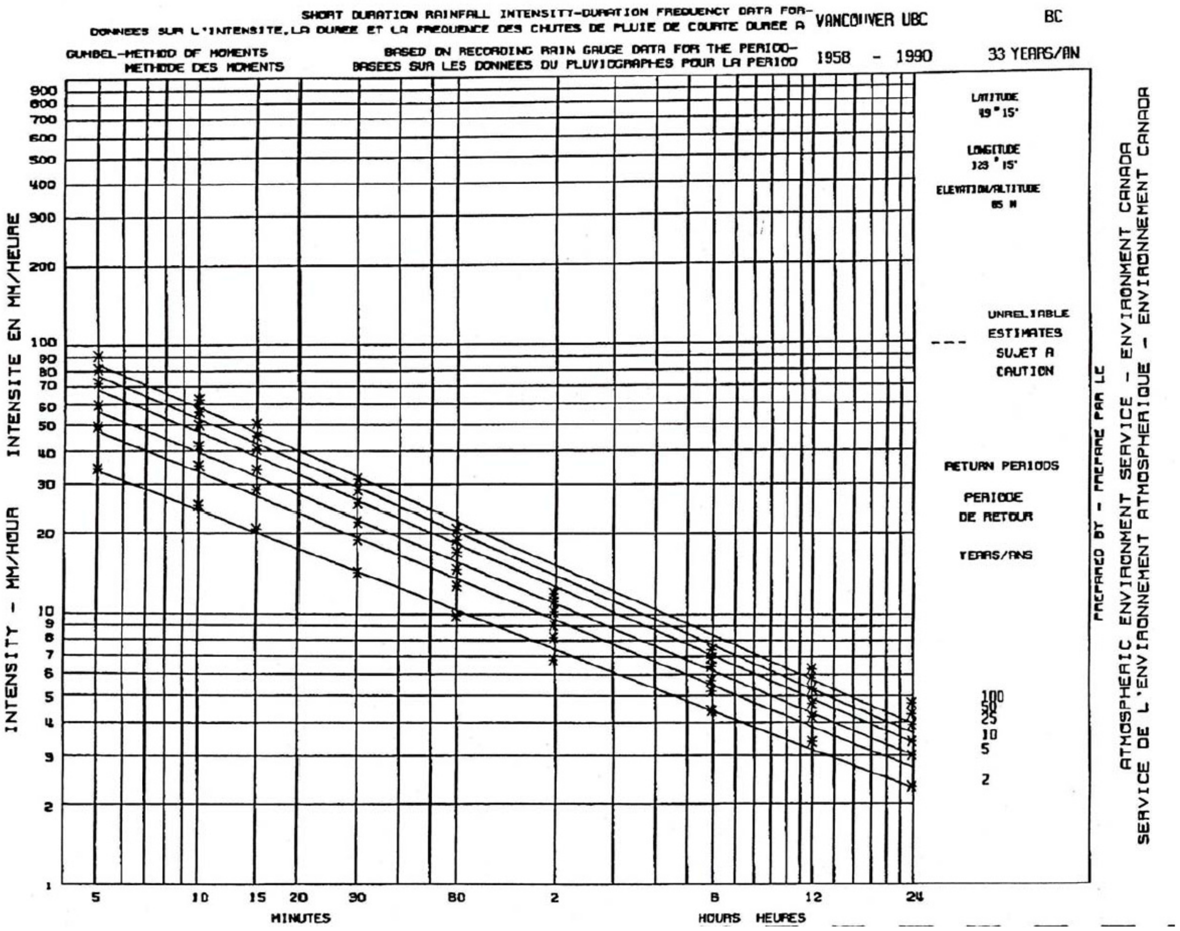
		T_R (yrs)	2			5			10		
		i (mm/hr)	36			48			55		
plot	**plot area** (m^2)	**runoff area** (m^2)	**rainfall** (L/s)	**runoff** (L/s)	**total** (L/s)	**rainfall** (L/s)	**runoff** (L/s)	**total** (L/s)	**rainfall** (L/s)	**runoff** (L/s)	**total** (L/s)
1	3.66	17.7	0.037	0.168	0.205	0.049	0.224	0.273	0.056	0.257	0.313
2	3.57	17.9	0.036	0.170	0.205	0.048	0.226	0.274	0.055	0.259	0.314
3	3.62	15.8	0.036	0.150	0.186	0.048	0.200	0.248	0.055	0.229	0.284
4	3.62	18.2	0.036	0.173	0.209	0.048	0.231	0.279	0.055	0.265	0.320
5	3.78	18.5	0.038	0.176	0.214	0.050	0.234	0.285	0.058	0.269	0.326
6	3.74	17.2	0.037	0.163	0.201	0.050	0.218	0.268	0.057	0.250	0.307
7	3.51	18.7	0.035	0.177	0.212	0.047	0.236	0.283	0.054	0.271	0.325
8	3.69	18.9	0.037	0.180	0.216	0.049	0.239	0.289	0.056	0.274	0.331
9	3.72	20.1	0.037	0.191	0.228	0.050	0.254	0.304	0.057	0.291	0.348
avg:	**3.66**	**18.1**	**0.037**	**0.172**	**0.209**	**0.049**	**0.229**	**0.278**	**0.056**	**0.263**	**0.319**

		T_R (yrs)	25			50			100		
		i (mm/hr)	67			72			81		
plot	**plot area** (m^2)	**runoff area** (m^2)	**rainfall** (L/s)	**runoff** (L/s)	**total** (L/s)	**rainfall** (L/s)	**runoff** (L/s)	**total** (L/s)	**rainfall** (L/s)	**runoff** (L/s)	**total** (L/s)
1	3.66	17.7	0.068	0.313	0.381	0.073	0.336	0.410	0.082	0.378	0.461
2	3.57	17.9	0.066	0.316	0.382	0.071	0.339	0.411	0.080	0.382	0.462
3	3.62	15.8	0.067	0.278	0.346	0.072	0.299	0.372	0.081	0.337	0.418
4	3.62	18.2	0.067	0.322	0.390	0.072	0.347	0.419	0.081	0.390	0.471
5	3.78	18.5	0.070	0.327	0.398	0.076	0.352	0.427	0.085	0.396	0.481
6	3.74	17.2	0.070	0.304	0.374	0.075	0.327	0.402	0.084	0.368	0.452
7	3.51	18.7	0.065	0.330	0.395	0.070	0.355	0.425	0.079	0.399	0.478
8	3.69	18.9	0.069	0.334	0.403	0.074	0.359	0.433	0.083	0.404	0.487
9	3.72	20.1	0.069	0.355	0.424	0.074	0.382	0.456	0.084	0.429	0.513
avg:	**3.66**	**18.1**	**0.068**	**0.320**	**0.388**	**0.073**	**0.344**	**0.417**	**0.082**	**0.387**	**0.469**


SHORT DURATION RAINFALL INTENSITY-DURATION FREQUENCY DATA FOR-
DONNEES SUR L'INTENSITE,LA DUREE ET LA FREQUENCE DES CHUTES DE PLUIE DE COURTE DUREE A
VANCOUVER UBC
BC
GUMBEL-METHOD OF MOMENTS
METHODE DES MOMENTS
BASED ON RECORDING RAIN GAUGE DATA FOR THE PERIOD-
BASEES SUR LES DONNEES DU PLUVIOGRAPHES POUR LA PERIOD
1958 - 1990
33 YEARS/AN
LATITUDE
49° 15'
LONGITUDE
123° 15'
ELEVATION/ALTITUDE
85 M
UNRELIABLE
ESTIMATES
SUJET A
CAUTION
RETURN PERIODS
PERIODE
DE RETOUR
YEARS/ANS
100
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25
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5
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INTENSITY - MM/HOUR INTENSITE EN MM/HEURE
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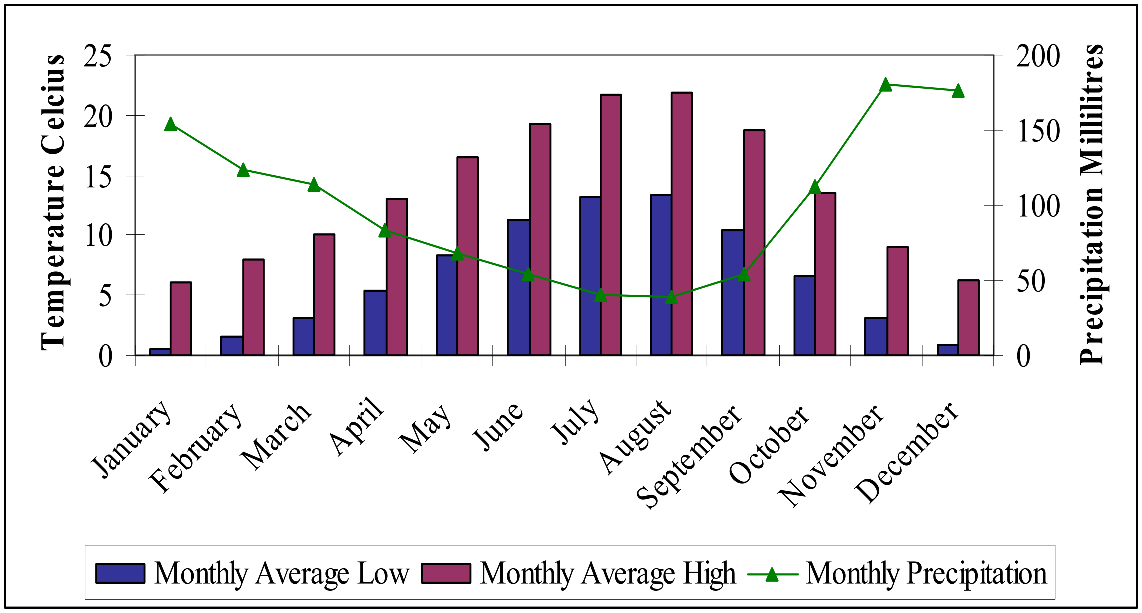


Figure 32- Climate Data for GVRD (Source-Environment Canada)

***Climatological information is based on monthly averages for the 30-year period 1971 - 2000**

***Mean number of precipitation days = Mean number of days with at least 0.2 mm of precipitation**

***Precipitation includes both rain and snow**

Table 16- Greater Vancouver Area, British Columbia Climate Data

Temperature:	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Daily Average (°C)	3.6	4.9	6.6	9.1	12.3	14.7	16.9	17.1	14.5	10.3	6.1	3.8
Standard Deviation	1.9	1.6	1.4	1.1	1	1	0.9	0.9	1	0.9	1.7	1.7
Daily Maximum (°C)	6	7.6	9.5	12.3	15.7	18	20.4	20.5	17.9	13.1	8.5	6.1
Daily Minimum (°C)	1.2	2.3	3.6	5.7	8.8	11.4	13.2	13.5	11.1	7.5	3.7	1.5
Extreme Maximum (°C)	16.4	17.9	19.5	25	31	30	31.1	**32.8**	30	25	17.5	14.6
Date (yyyy/dd)	1981/21	1986/27	1992/30	1976/30+	1983/29	1970/02	1961/12	**1960/09**	1973/05	1975/01	1980/04	1980/25
Extreme Minimum (°C)	-13.9	-12.5	-10.6	-0.6	1.7	2.6	7.3	6.1	0.6	-4.1	-12.5	**-18.3**
Date (yyyy/dd)	1969/28	1989/02	1971/01	1976/01	1965/06	1983/27	1979/01+	1973/18	1961/29	1984/31	1985/27+	**1968/29**
Rainfall (mm)	146.5	125.2	118.7	89	68.3	55.5	39.3	48.1	58.6	113.3	196.1	167.9
Snowfall (cm)	16.1	12.3	3.1	0.6	0	0	0	0	0	0.3	2.8	15.6
Precipitation (mm)	162.7	137.5	121.9	89.6	68.3	55.5	39.3	48.1	58.6	113.6	198.9	183.5
Average Snow Depth (cm)	1	1	0	0	0	0	0	0	0	0	0	1
Median Snow Depth (cm)	0	0	0	0	0	0	0	0	0	0	0	1
Snow Depth at Month-end (cm)	0	0	0	0	0	0	0	0	0	0	0	2
Extreme Daily Rainfall (mm)	**88.1**	53.4	64.5	41.2	36.1	40.8	56.9	54	44.9	72.9	55.4	85.1
Date (yyyy/dd)	**1968/18**	1979/24	1976/23	1983/09	1974/24	1992/28	1972/11	1991/29	1986/22	1975/16	1964/29	1972/25
Extreme Daily Snowfall (cm)	23.6	24.4	17.8	10	0	0	0	0	0	5.2	14	**30**
Date (yyyy/dd)	1971/12	1989/17	1974/07	1981/11	1958/01+	1958/01+	1958/01+	1958/01+	1957/01+	1984/31	1975/29	**1990/29**
Extreme Daily Precipitation (mm)	**88.1**	53.4	64.5	41.2	36.1	40.8	56.9	54	44.9	72.9	55.4	85.1
Date (yyyy/dd)	**1968/18**	1979/24	1976/23	1983/09	1974/24	1992/28	1972/11	1991/29	1986/22	1975/16	1964/29	1972/25
Extreme Snow Depth (cm)	**51**	22	7	10	0	0	0	0	0	0	7	33
Date (yyyy/dd)	**1971/15**	1986/16	1989/02	1981/12	1971/01+	1971/01+	1971/01+	1971/01+	1971/01+	1971/01+	1985/30	1971/14
<= 0 °C	1.6	0.28	0.09	0	0	0	0	0	0	0	0.52	1.8
> 0 °C	29.4	28	30.9	30	31	30	31	31	30	31	29.5	29.2
> 10 °C	2.9	5.7	11.4	22.5	30.6	30	31	31	29.9	26.9	8.1	2.7
> 20 °C	0	0	0	0.44	3.1	6	15.4	16.7	5.9	0.43	0	0
> 30 °C	0	0	0	0	0.04	0	0	0	0	0	0	0
> 35 °C	0	0	0	0	0	0	0	0	0	0	0	0
> 0 °C	20.2	21.6	27	29.7	31	30	31	31	30	30.7	25.9	20.4
<= 2 °C	16.9	12.8	8.2	1.6	0	0	0	0	0.04	0.79	8.2	16.4
<= 0 °C	10.8	6.7	2.7	0.28	0	0	0	0	0	0.29	4.1	10.5
< -2 °C	6.1	2.8	0.46	0	0	0	0	0	0	0.13	1.9	4.6
< -10 °C	0.12	0.08	0.04	0	0	0	0	0	0	0	0.13	0.29

APPENDIX B – Background on Humic Substances

Classification of Humic Substances:

Classically, humic substances have been divided into three main groups based on their solubilities in acids and bases. Humic acids are soluble in base but insoluble in acid, while fulvic acids are soluble in both acids and bases. Humin, the third fraction of humic substances, is neither soluble in acids or bases (Tipping, 2002). In general, humic acids and humins are found as part of the natural soil solid phase, while fulvic acids are largely found as accounting for the large fraction of dissolved organic matter in natural waters (Tipping, 2002).

Formation of Humic Substances:

Humic substances are mainly derived from terrestrial plant material but other contributions may be from animal and microbial remains and from microbial synthesized products (Tipping, 2002).

Decomposition of Humic Substances:

Humic substances are able to resist degradation due to their chemical and physical heterogeneity (MacCarthy and Rice, 1991). Humic substances tend to be less susceptible to break down when they are adsorbed to minerals, aggregated, or when complexed with metal ions (McKeague et al, 1986).

Optical Properties of Humic Substances:

Cation-humic interactions control the reactivity of the cation, including its bioavailability (Tipping, 2002). The cation also influences the physico-chemical nature of the humic and thus it is important to consider interactions between cations and the humics it is bound to. In general, metals bound to humic substances are less bio-available than ionic metals (Grimes et al., 1999).

Humic substances resist degradation. One such proposition includes the fact that there are no enzymes capable of degrading the compounds due to the chemical and physical heterogeneity of the substance. Humic substances tend to be less susceptible to breakdown when they are adsorbed by minerals, when aggregated or when complexed with metal ions (Tipping, 2002).

Approximately 80% of hydrogen in humic matter is associated to carbon, the rest to oxygen. Since hydrogen is only able to dissociate from oxygen, an upper limit of 10 meg g-1 of protons can dissociate (Tipping, 2002). The proton dissociating groups are the most important groups for binding of cations (Tipping, 2002). Total content for proton dissociating groups in fulvic acids ranges from 6 to 10 meg g-1 while for humic acids the range is from 4 to 6 meg g-1 (Tipping, 2002). The most prevalent proton dissociating groups at $pH < 7$ are carboxylic. Nitrogen and sulfur may also be site of strong cation binding (Tipping, 2002).

Additional Isotherm Models:

Langmuir Isotherm:

The Langmuir isotherm was developed for the sorption of gases onto solids assuming that the attractive forces of the unsaturated binding sites only extend as far the diameter of one sorbed molecule, thus only forming a monolayer on the sorbate (Volesky, 2003). It is also assumed that the sorbed species do not interact with each other and the species in solution do not react as well. The Langmuir Isotherm has the following relationship:

$$q_e = \frac{Q_T K C_e}{1 + K C_e} \qquad [8]$$

Where,

q_e = uptake of sorbate;
Q_T = total binding capacity of the sorbent;
K = apparent equilibrium cons tan t;
and
C_e = equilibrium solution concentration of sorbate.

The Langmuir isotherm is widely used because of its simplicity and has been used for empirical modeling of many sorption systems where the mechanisms are not known (Volesky, 2003). The major limitation with this model is that the binding constants and the capacity are determined for the species of interest without specific reference to important solution chemistry such as pH, ionic strength and the concentration of competing ions. Thus, this model is only capable of confidently predicting sorption behavior over a small range of conditions.

Freundlich Isotherm:

The Freundlish Isotherm is an empirical exponential isotherm that was developed for activated carbon and assumes surface binding (Volesky, 2003). The major limitation of this model is that it has an infinite binding capacity, which limits its applicability to solutions of low to moderate sorbate concentrations (Volesky, 2003). However, due to its mathematical simplicity, the Freudlich isotherm is commonly used in more complex sorption modeling (i.e. sorption performance modeling) (Volesky, 2003). The Freundlich relationship can be found below as:

$$q_e = kC_e^{(1/n)} \qquad [9]$$

Where,

$\text{k and n are Freundlich constants.}$

Brunauer, Emmett and Teller Model (BET):

The BET model assumes that multiple layers form on an adsorption surface and that each layer can be described by the Langmuir isotherm. The model also assumes that the given layer does not need to be completed before subsequent layers begin to form (Volesky, 2003). The mathematical relationship for the BET model can be found below:

$$q_e = \frac{BQC_e}{(C_s - C_e)[1 + (B-1)(C_e / C_s)]} \qquad [10]$$

Where,

$C_s = \text{saturation constant of the solute;}$

$B = \text{surface interaction energy constant;}$

and

$Q = \text{moles of sorbate sorbed per mass of sorbent to complete a monolayer.}$

Several models for proton binding on humic substances exist which may account for competition and variations in ionic strengths. The most recent and advanced models are Model VI and the NICCA (consistent non-ideal competitive adsorption) model (Tipping, 2002).

APPENDIX C- Metals and Vegetation

This section discusses the toxicity of Zn, Cu, Cd, Pb, Cr, and Fe on vegetation.

C.1 Zinc

Zinc (Zn) is supposed to be the least toxic of the heavy metals to plants. It is an essential metal to higher plants and is involved in several important metabolic processes. Zn also plays a role in protein synthesis, the electron transport system, and forms stable complexes with DNA and RNA therefore it may also affect DNA and RNA stability. (Collins, 1981) Both Zn deficiency and toxicity cause significant changes in the metabolism of plants resulting in growth retardation, stunted growth and chlorosis affecting both the roots and shoots (Pahlsson, 1989). Further, the epidermis of roots and the cells in the epidermis may become lignified (Paivoke, 1983). Usually a concentration of at least 1000 µg/L of Zn is required to affect growth under laboratory conditions with nutrient solutions. A concentration of 2000 µg/L of Zn decreases the shoot and root growth of ryegrass (Pahlsson, 1989). Several grass species are capable of evolving tolerance to Zn, but not necessarily to other heavy metals. The addition of humic acid as a complexing agent may reduce toxicity compared to if only ionic Zn is present in the growth medium (Marquenie-Van der Werf et al., 1981).

Very few studies have been devoted to the effect of heavy metals on reproduction and seed germination and seedling growth. Usually very high Zn concentrations have been used in the growth medium and the results are somewhat conflicting. The seed germination itself seems to be rather insensitive to Zn and also different pH. At Zn levels lower than 10^4 µg/L in the germination medium, seed germination from both tree and grass species has proved to be unaffected (Jordan, 1975; Scherbatskoy et al., 1987). However, the further development and survival of the seedlings was not followed up. The concentrations of Zn in highway runoff are substantially less (10-910 µg/L of Zn (Table 3) than the 1000 µg/L of Zn is required to affect growth therefore acute or chronic Zn toxicity is not expected under the experimental conditions proposed.

C.2 Copper

Copper (Cu), like Zn, is an essential micronutrient for normal plant metabolism. Cu is involved as a key component of several enzymes involved in carbohydrate, nitrogen and cell wall metabolism. Further, Cu is important to seed production, disease resistance and the water relations in the plant (Bussler, 1981). The amount of Cu necessary for normal growth and development are generally supplied in the soil. At higher soil levels, Cu can be extremely toxic, although several species are able to evolve tolerant genotypes. However, like other heavy metals, the degree of toxicity is dependent on multiple parameters including the nutritional status of the plant. As an example, phosphorus deficiency in the plant may enhance the toxicity of Cu (Wallace, 1984). Generally visible symptoms of Cu are small chlorotic leaves and early leaf fall. Further, the growth is stunted and initiation of roots and development may result in a lower water and nutrient uptake leading to disturbances in the metabolism and growth retardations (Pahlsson 1989).

There is a great difference in response between and within species grown at fairly similar growth conditions which complicates attempts to set a concentration limit in a culture solution at which Cu may be toxic to plants. In different soil media, additional factors must be considered, including availability and uptake conditions (Pahlsson 1989). Therefore, leaf tissue concentrations of a metal would be a better measurement tool. Critical leaf tissue concentrations of Cu are limited to a comparatively narrow interval in which the metal is shown to be toxic. Generally most species are effected between 15 and 25 μg/g Cu dry weight, however corresponding concentrations of Cu in culture solutions are shown to vary considerably (Beckett and Davis, 1977; Davis et al., 1978; MacNicol and Beckett, 1985). The concentrations of Cu in highway runoff are substantially low (14-220 μg/L of Cu (Table 3) however, with the variability associated with Cu toxicity, no concrete predictions can be made concerning the affect Cu will have on the proposed experiments.

C.3 Cadmium

The toxicity of Cadmium (Cd) to plants is well documented. Chemically Cd is similar to Zn and available Cd in the growth medium is easily taken up by the plants. Cd is not known to be an essential element to plants. Although a limited transport of Cd to shoots and binding to cell walls occur in the roots, Cd is relatively more toxic to plants than Lead (Pb). The strong affinity of Cd ions for sulhydryl groups of several compounds and phosphate groups involved in plant metabolism might explain the great toxicity (Pahlsson 1989). Plants treated with high (122- 8000 µg/L Cd, dependent on species) concentrations of Cd usually become stunted in growth. The leaves are smaller, curled and chlorotic and leaf margins and veins show a red-brown colouration (Pahlsson 1989). According to Beckett and Davis (1977) the critical tissue concentration of Cd, at which the metal causes a biomass decrease, is fairly independent of growth conditions. Although the variation between species is great, the lower limit to affect growth ranges from 3 -10 µg/g Cd dry weight of leaf or shoot tissue. When applying Cd-salt solutions, biomass production was fairly unaffected, but the length of aboveground shoots was significantly reduced. The concentrations of Cd in highway runoff are substantially less (4-10 µg/L of Cd (Table 3) than the 100+ µg/L of Cd is required to affect growth therefore acute or chronic Cd toxicity is not expected under the experimental conditions proposed.

C.4 Lead

Similarly to Cd, lead (Pb), is considered a nonessential metal to plants. Compared to Cd, the phytotoxicity of Pb is relatively low. Nutrient solutions containing 100- 200 µg/L Pb are shown to affect root growth and elongation as well as affect an enzyme involved in photosynthesis and root protein content. Cytological disturbances in root cells are observed at 600 µg/L Pb or more (Pahlsson 1989). Both physiological and biochemical processes are affected by an excess of Pb. As the metal reacts with important functional groups, the activity of several enzymes is influenced, some important in the photosynthesis and nitrogen metabolism (Pahlsson 1989). Visible symptoms of toxicity, similar to other heavy metals, are smaller leaves and stunted growth. Leaves may become chlorotic and reddish with necrosis and the roots turn black.

In most studies, generally only very high levels of Pb in the growth medium have proved to affect growth in plants. As with other heavy metals, the toxicity to seed germination is also quite small with Pb. The concentrations of Cd in highway runoff are in the range (7.4-560 μg/L of Pb (Table 3) of the concentration Pb required to affect growth. Therefore acute or chronic Pb toxicity may occur under the experimental conditions proposed.

C.5 Chromium

Chromium is a highly toxic non-essential metal for microorganisms and plants. Due to its widespread industrial use, chromium (Cr) has become a serious pollutant in the environment. The hexavalent form of the metal, Cr(VI), is considered a more toxic species than the relatively innocuous and less mobile Cr(III) form (Cervantes et all 2001). Cr(VI) has been demonstrated to produce serious damage to living cells, it is considered that Cr(III) is less toxic because of its extremely low solubility, which prevents its leaching into ground water or its uptake by plants. However, studies in plants have shown that Cr(III) also produces serious problems in living tissues although at higher concentrations than Cr(VI). Toxic symptoms produced by Cr(VI) are stronger than those caused by Cr(III), and occur earlier and at lower concentrations.

Since plants lack a specific transport system for Cr, it is taken up by carriers of essential ions such as sulfate or iron. Toxic effects of Cr on plant growth and development include alterations in the germination process as well as in the growth of roots, stems and leaves, which may affect total dry matter production and yield. Cr also causes deleterious effects on plant physiological processes such as photosynthesis, water relations and mineral nutrition. Metabolic alterations by Cr exposure have also been described in plants either by a direct effect on enzymes or other metabolites or by its ability to generate reactive oxygen species which may cause oxidative stress.

C.6 Iron

Iron (Fe) is essential for many plant functions including chlorophyll development and function, energy transfer, component in enzymes and proteins, plant respiration and plant metabolism and is involved in nitrogen fixation (Anderson, 2006). Fe toxicity is primarily pH related and occurs where the soil pH has dropped sufficiently ($pH < 5.0$) to create an excess of available Fe. The visible symptoms of Fe toxicity are likely to be a deficiency of another nutrient. Fe toxicity can also occur when Zn is deficient or the soil is in a "reduced" condition caused by very wet or flooded conditions. Excess Fe can result in dark green foliage, stunted growth of tops and roots, dark brown to purple leaves in some plants (Anderson, 2006). The concentrations of Fe in highway runoff (1600-9050 μg/L of Fe (Table 3) are not expected to affect than growth of vegetation because it is an essential element. Therefore acute or chronic Fe toxicity is not expected under the experimental conditions proposed. Zn deficiency will not be an issue and the compost in the treatment system acts as a buffer for pH.

APPENDIX D – Experimental Procedures

Moisture Content

The compost moisture content may be expressed by weight as the ratio of the mass of water present to the dry to the dry weight of the compost sample, or by volume as ratio of volume of water to the total volume of the compost sample. To determine any of these ratios for a particular compost sample, the water mass must be determined by drying the soil to constant weight and measuring the soil sample mass after and before drying. The water mass (or weight) is the difference between the weights of the wet and oven dry samples. The criterion for a dry soil sample is the soil sample that has been dried to constant weight in oven at temperature between 100 – 110 °C. This temperature range is based on water boiling temperature and does not consider the soil physical and chemical characteristics. The procedure used to determine the compost moisture content is outlined below:

1. Three compost samples of about 10g were weighed and recorded as wet compost.
2. Samples were placed in the oven 105°C and dried for 24 hours.
3. Samples were then weighed and recorded as dry compost.
4. Samples were returned to the oven for an additional 24 hours and weighed again.
5. Step 4 was repeated until there was no difference between any two consecutive measurements of the weight of dry compost.

Effective Porosity

The effective porosity, p_e, also called the kinematic porosity, of a porous medium is defined as the ratio of the part of the pore volume where the water can circulate to the total volume of a representative sample of the medium. In naturally porous systems such as subsurface soil, where the flow of water is caused by the composition of capillary, molecular, and gravitational forces, the effective porosity can be approximated by the specific yield, or drainage porosity, which is defined as the ratio of the volume of water drained by gravity from a saturated representative sample of the soil to the total volume of the sample. The effective porosity, or open porosity, the compost media refers to the fraction of the total volume in which fluid flow is effectively taking place. This excludes dead-end pores or non-connected cavities and can be determined by following a standard test method (TMECC 2002). The effective porosity (percentage of soil that is air) of the compost was determined using the method below:

1. Three compost samples of approximately 400g were weighed out.
2. Samples were transferred to a graduated cylinder and the volume was recorded.
3. Samples were saturated with water and then allowed to drain completely under the action of gravity until it gets to its irreducible saturation.
4. A measured volume of water is added until the compost sample is covered with water.
5. The effective porosity is represented by the known amount of water or the free air space (FAS).

Bulk Density

Density (g/ml) =Weight of Soil / Volume of Soil

Dry Root Mass

The dry root mass percentage in the compost media (at seven months of growth) was calculated using the following procedure:

1. Three compost plus root samples of approximately 100g were weighed out.
2. Roots were separated from compost.
3. Compost and roots were placed in a drying oven (at 200 °C) for 48 hours.
4. Compost and roots were weighed after drying and a ratio obtained.

APPENDIX E – Experimental Apparatus

Calibration Curves for Peristaltic Pumps Used in Experimentation

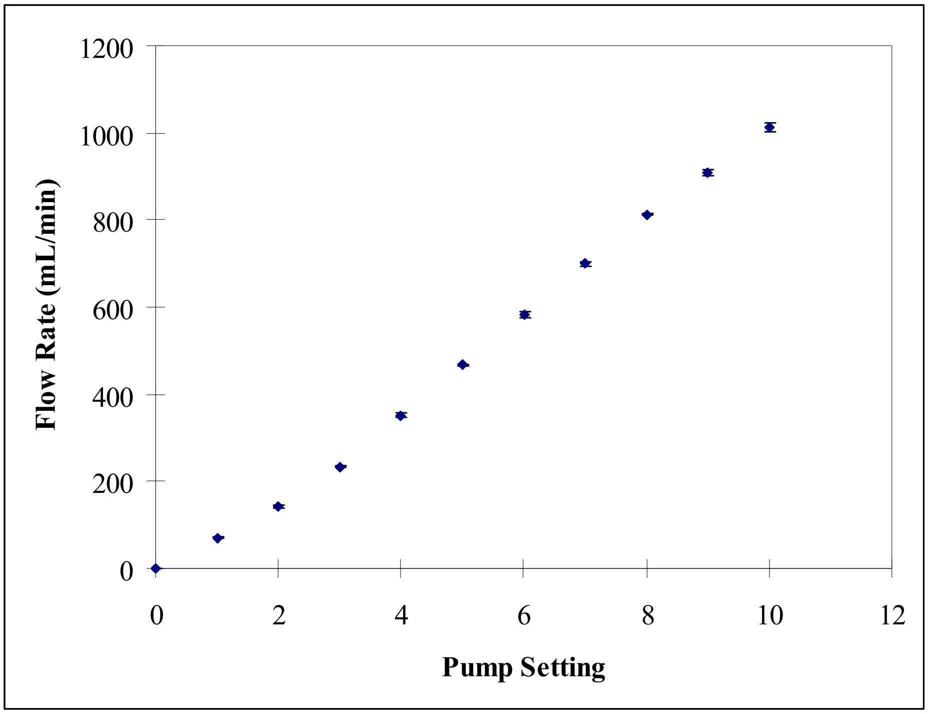


Figure 33- Peristaltic Pump (Model 3640) Flow Rate Capabilities, n=3, error bars= one standard deviation

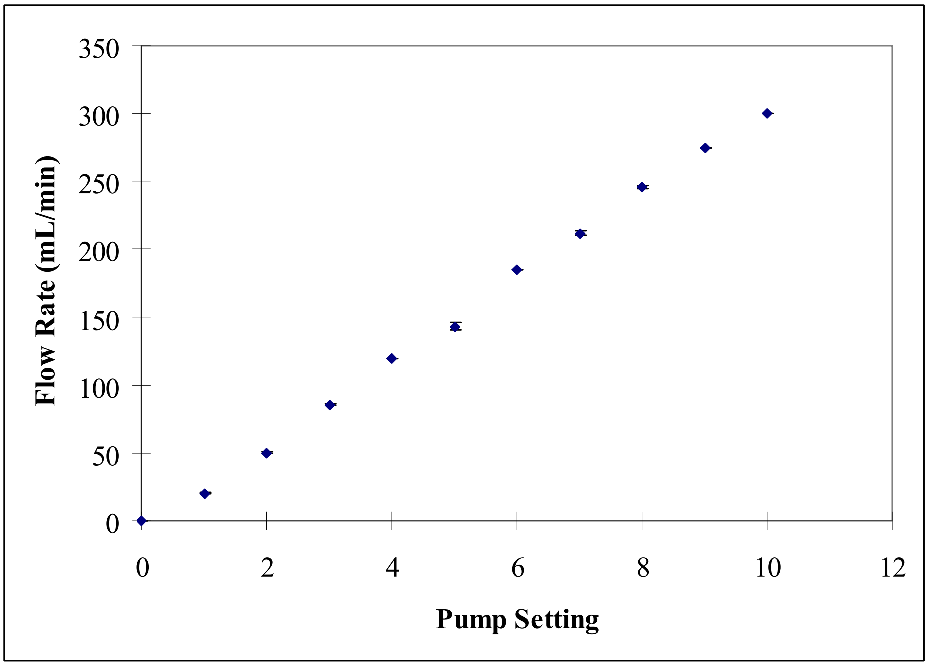


Figure 34- Peristaltic Pump (Dual Head) Flow Rate Capabilities, n=3, error bars= one standard deviation

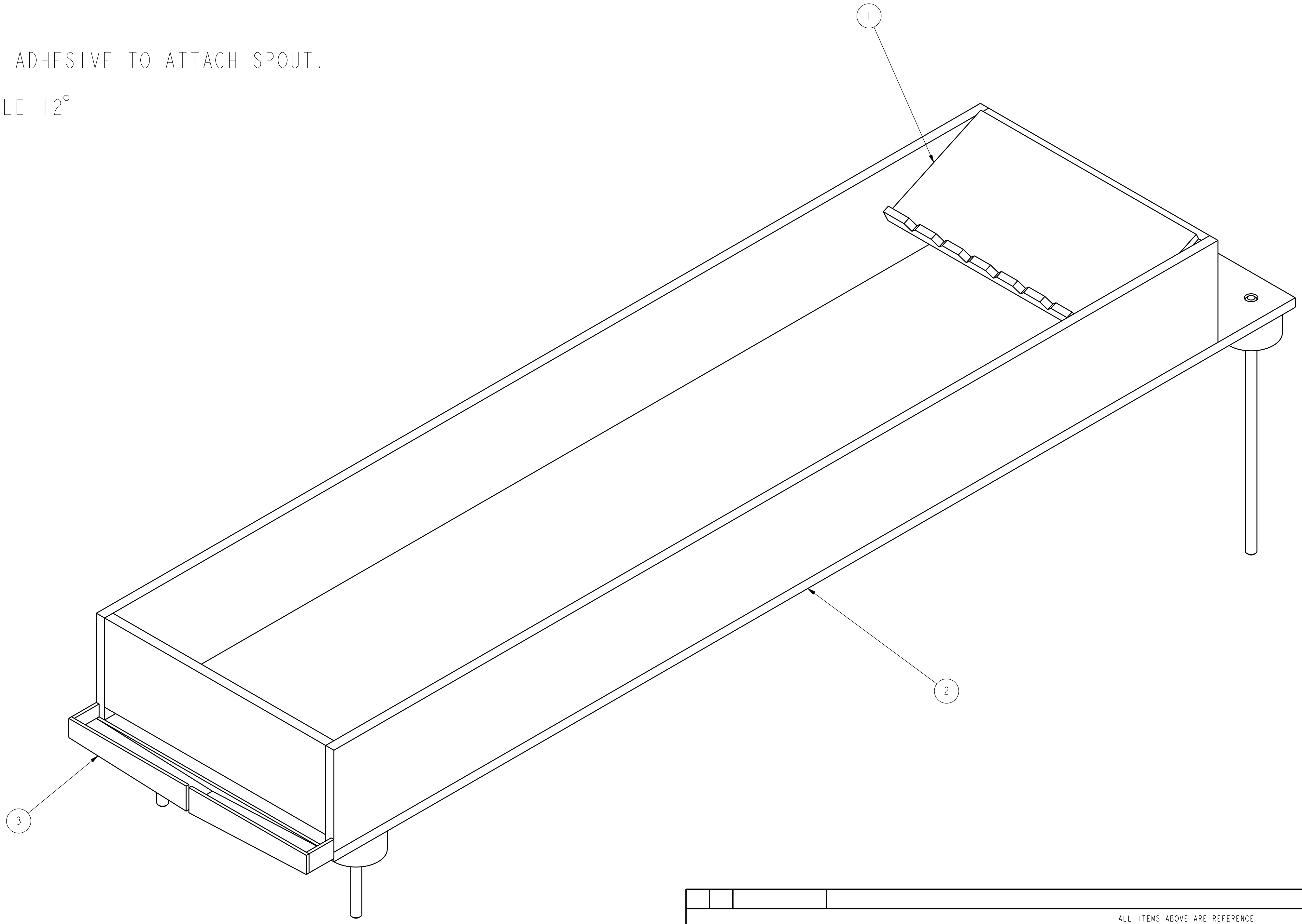
INSPECTION POINT	TOTAL INSP. PTS: -

PART NO	SH	REV
-	1	1

NOTES:

1) USE APPROPRIATE ADHESIVE TO ATTACH SPOUT.

2) OPTIMUM BED ANGLE 12°



ALL ITEMS ABOVE ARE REFERENCE								
3	1	-	TEST BED SPOUT/SAMPLE PORT	BASE_BED_SPOUT	-		-	
2	1	-	TEST BED BASE UNIT	BASE_BED	-		-	
1	1	-	FLOW CHARACTERIZATION BED 7 WEIR FLOW DISTRIBUTOR	7_WEIR_FLOW_DIST	-		-	
ITEM NO	QTY REQD	MATERIAL USAGE OR COMPONENT NUMBER	NOMENCLATURE OR DESCRIPTION	OWC NO	PART NO	CAGE CODE	CERT REQ	NOTE NO

DRAWING BILL OF MATERIALS

TOLERANCES
X.X ± 0.025 FRACTIONS ± 1/16
X.XX ± 0.010 ANGLES ± 0.5°
X.XXX ± 0.003

U.N.O. MACH. SURFACE FINISH 125
BREAK SHARP EDGES: 0.005 TO 0.010
FILLET RADII: 0.005 TO 0.010

THIRD ANGLE PROJECTION

DRAWING IN ACCORDANCE WITH ANSI Y14.5M

INCHES

DIMENSIONS ARE IN INCHES. DIMENSIONS SHOWN IN BRACKETS ARE MILLIMETERS.

-	-
-	-
-	-
-	-
-	-
-	-
-	-
NEXT ASSEMBLY	USED ON

APPLICATION

EXPERIMENTAL APPARATUS

FLOW CHARACTERIZATION BED

FULL ASSEMBLY

SIZE	CAGE CODE	PART NO:	REV
D		FLOW_CHAR_BED -	1

SCALE: 1:2	CALC WT: 0.000	ACTUAL WT: -	SHEET: 1 OF 1

INSPECTION POINT	TOTAL INSP. PTS: -

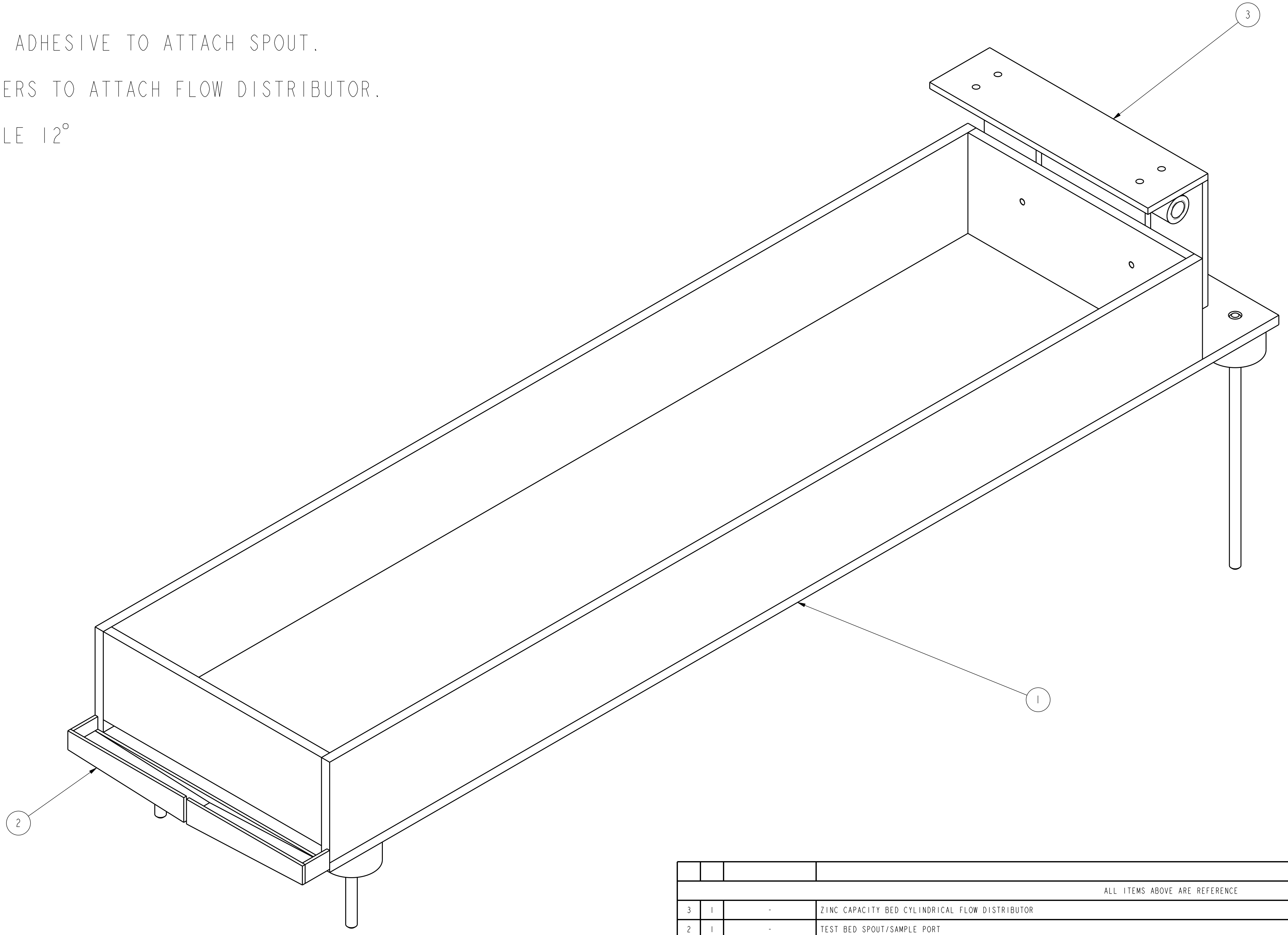
PART NO	SH	REV
-	1	1

NOTES:

1) USE APPROPRIATE ADHESIVE TO ATTACH SPOUT.

2) USE 1/4" FASTENERS TO ATTACH FLOW DISTRIBUTOR.

3) OPTIMUM BED ANGLE 12°



ALL ITEMS ABOVE ARE REFERENCE								
3	1	-	ZINC CAPACITY BED CYLINDRICAL FLOW DISTRIBUTOR	CYL_FLOW_DIST	-		-	
2	1	-	TEST BED SPOUT/SAMPLE PORT	BASE_BED_SPOUT	-		-	
1	1	-	TEST BED BASE UNIT	BASE_BED	-		-	
ITEM NO	QTY REQD	MATERIAL USAGE OR COMPONENT NUMBER	NOMENCLATURE OR DESCRIPTION	OWC NO	PART NO	CAGE CODE	CERT REQ	NOTE NO

DRAWING BILL OF MATERIALS

TOLERANCES

X.X ± 0.025 FRACTIONS ± 1/16

X.XX ± 0.010 ANGLES ± 0.5°

X.XXX ± 0.003

U.N.O. MACH. SURFACE FINISH 125

BREAK SHARP EDGES: 0.005 TO 0.010

FILLET RADII: 0.005 TO 0.010

THIRD ANGLE PROJECTION

DRAWING IN ACCORDANCE WITH ANSI Y14.5M

INCHES

DIMENSIONS ARE IN INCHES. DIMENSIONS SHOWN IN BRACKETS ARE MILLIMETERS.

-	-
-	-
-	-
-	-
-	-
-	-
-	-
NEXT ASSEMBLY	USED ON
APPLICATION	

EXPERIMENTAL APPARATUS

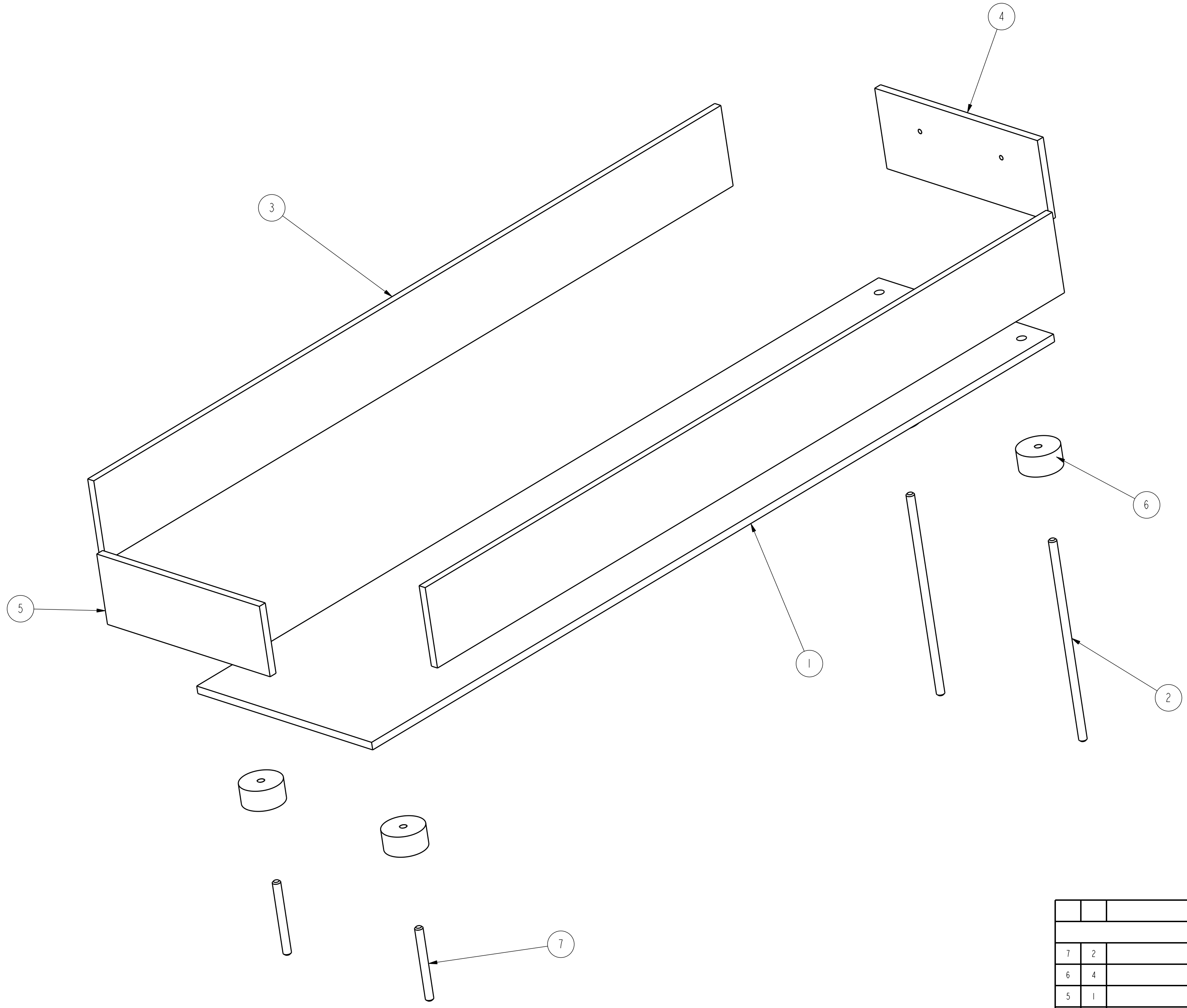
ZINC CAPACITY BED

FULL ASSEMBLY

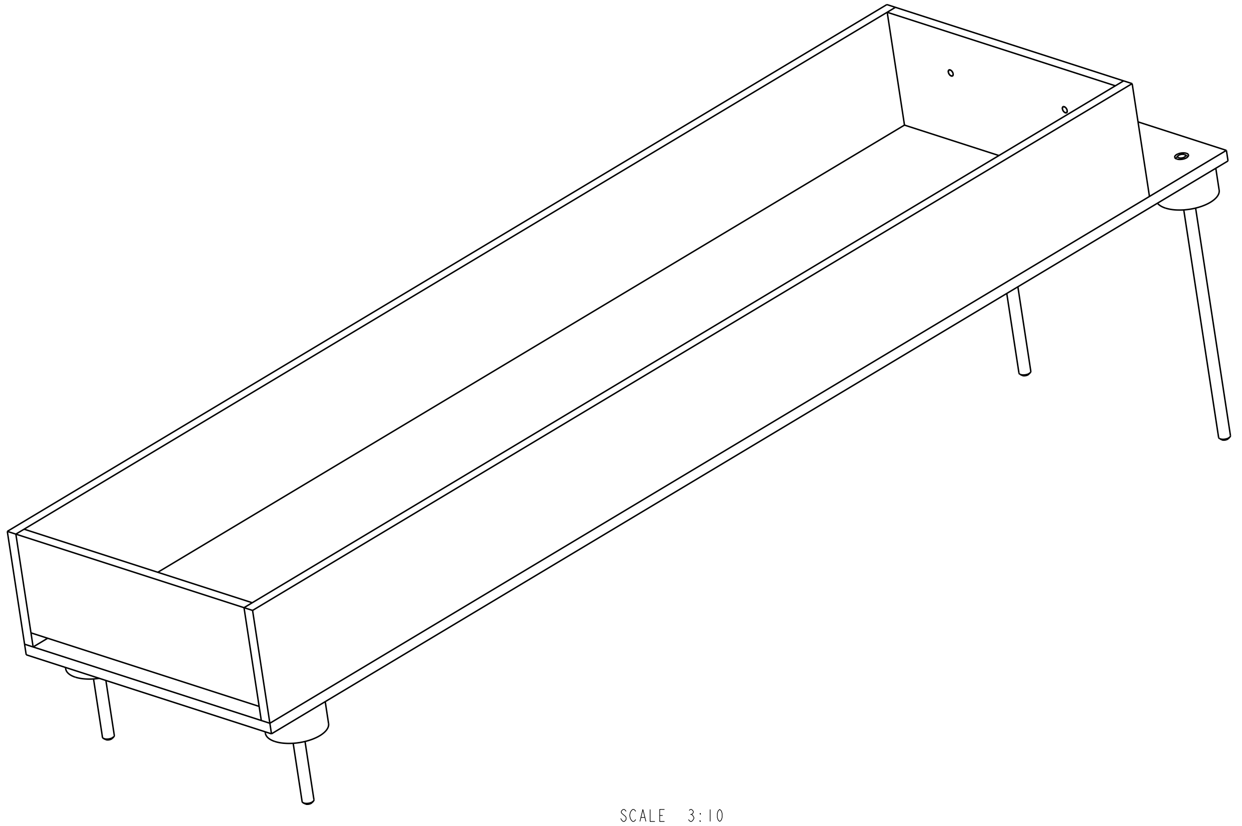
SIZE	CAGE CODE	PART NO:	REV
D		ZINC_CAP_BED -	1
SCALE: 1:2	CALC WT: 0.000	ACTUAL WT: -	SHEET: 1 OF 1

INSPECTION POINT	TOTAL INSP. PTS: n

PART NO	SH	REV
-	1	1



SCALE 3:10



SCALE 3:10

ALL ITEMS ABOVE ARE REFERENCE								
7	2		- (3/8 THREADED ROD CARBON STEEL - / -)	CAP_BED_LEG_FRT	-		-	
6	4		- (2 IN ROUND BAR ACRYLIC-CLEAR - / -)	CAP_BED-05	-		-	
5	1		- (3/8 SHEET ACRYLIC-CLEAR - / -)	CAP_BED-04	-		-	
4	1		- (3/8 SHEET ACRYLIC-CLEAR - / -)	CAP_BED-03	-		-	
3	2		- (3/8 SHEET ACRYLIC-CLEAR - / -)	CAP_BED-02	-		-	
2	2		- (3/8 THREADED ROD CARBON STEEL - / -)	BASE_BED_LEG_RR	-		-	
1	1		- (3/8 SHEET ACRYLIC-CLEAR - / -)	BASE_BED-01	-		-	
ITEM NO	QTY REQD	MATERIAL USAGE OR COMPONENT NUMBER	NOMENCLATURE OR DESCRIPTION	OWC NO	PART NO	CAGE CODE	CERT REQ	NOTE NO

DRAWING BILL OF MATERIALS

TOLERANCES

X.X ± 0.025 FRACTIONS ± 1/16

X.XX ± 0.010 ANGLES ± 0.5°

X.XXX ± 0.003

U.N.O. MACH. SURFACE FINISH 125

BREAK SHARP EDGES: 0.005 TO 0.010

FILLET RADII: 0.005 TO 0.010

THIRD ANGLE PROJECTION

DRAWING IN ACCORDANCE WITH ANSI Y14.5M

INCHES

DIMENSIONS ARE IN INCHES. DIMENSIONS SHOWN IN BRACKETS ARE MILLIMETERS.

-	-
-	-
-	-
-	-
-	-
-	-
-	-
NEXT ASSEMBLY	USED ON
APPLICATION	

EXPERIMENTAL APPARATUS

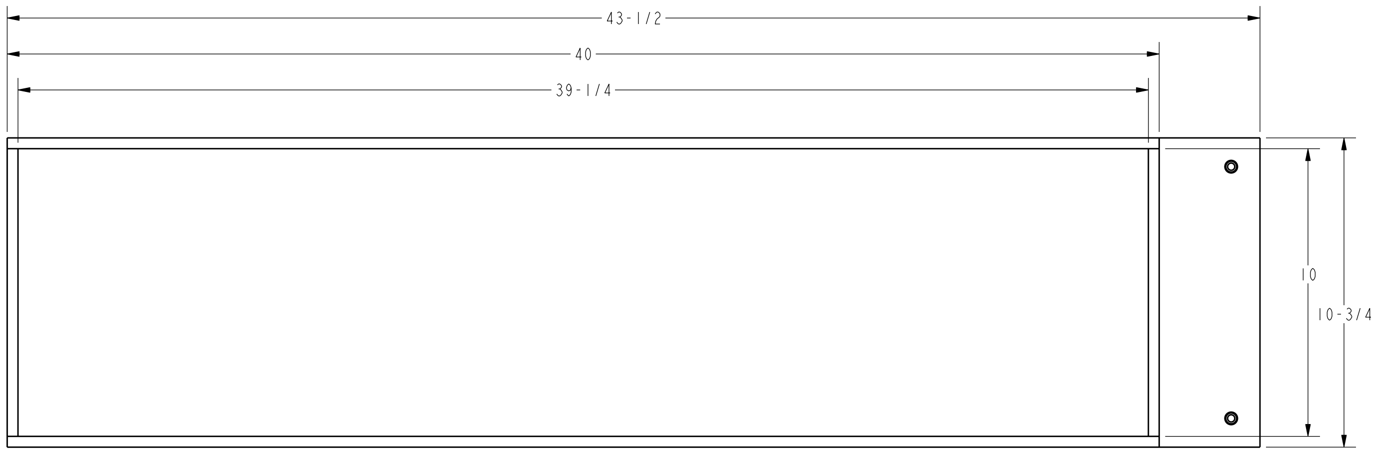
TEST BED

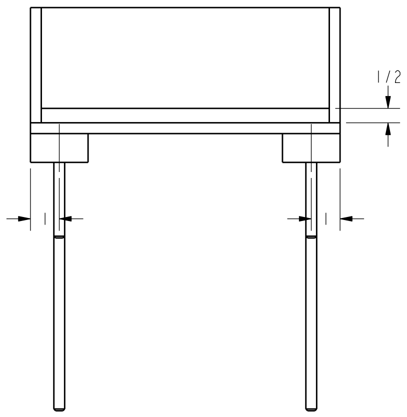
BASE UNIT

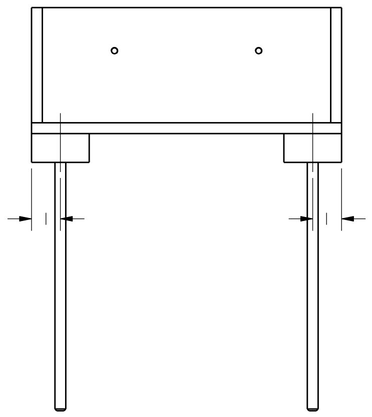
SIZE	CAGE CODE	PART NO:	REV
D	-	BASE_BED -	1
SCALE: 1:1	CALC WT: 0.000	ACTUAL WT: -	SHEET: 1 OF 3

NOTES:

1) USE APPROPRIATE ADHESIVE FOR ACRYLIC IN ASSEMBLY.

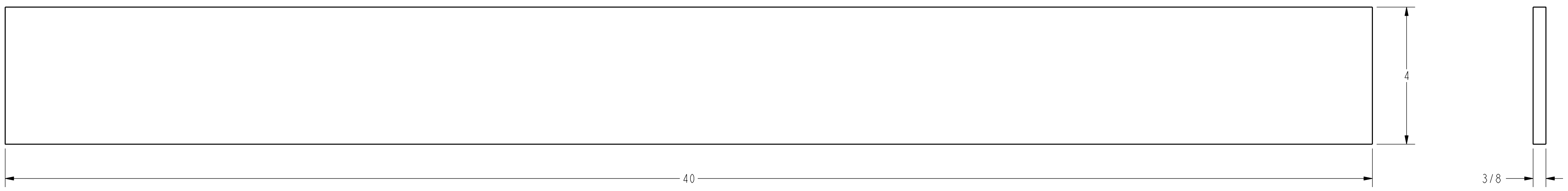




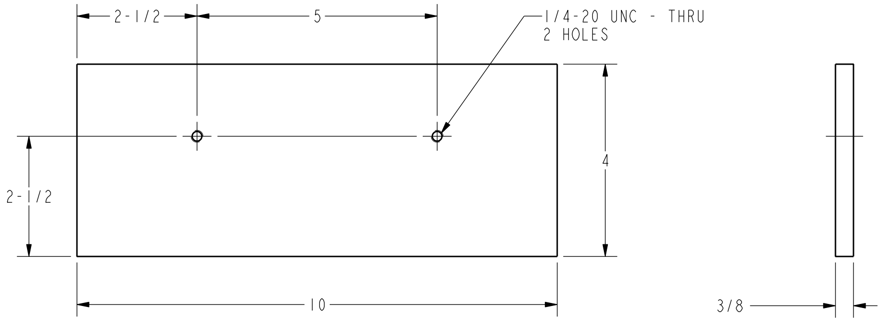


SIZE	CAGE CODE	OWC NO: BASE_BED	REV
D	-	PART NO: -	1
SCALE: 3:10	CALC WT: 0.000	ACTUAL WT: -	SHEETS: 2 OF 3

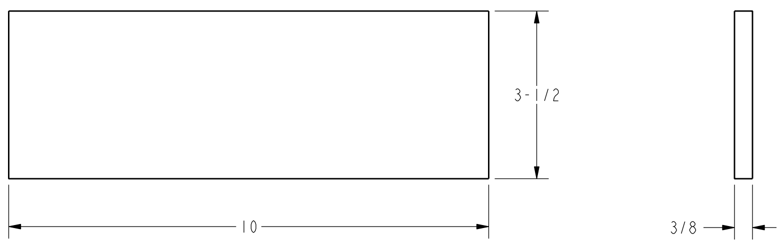
PART #1 - CAP_BED-01
SCALE 1:2



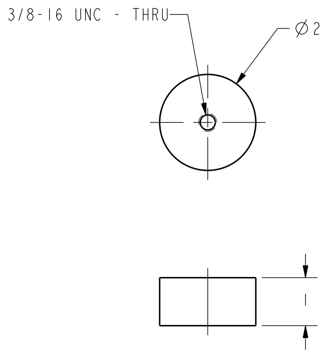
PART #2 - CAP_BED-02
SCALE 1:2



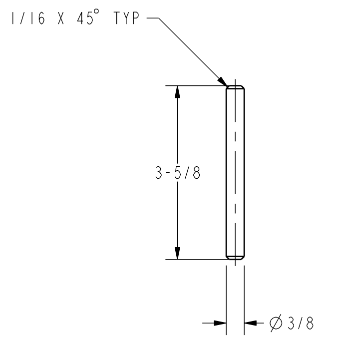
PART #3 - CAP_BED-03
SCALE 1:2



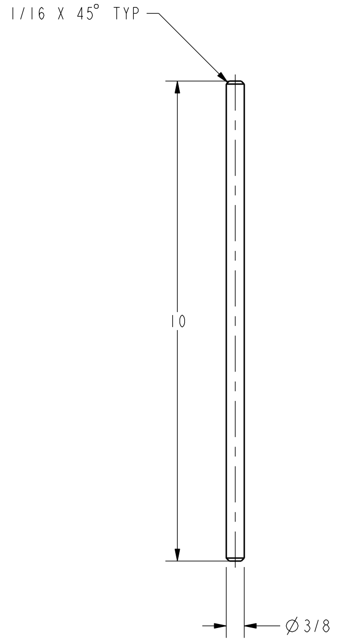
PART #4 - CAP_BED-04
SCALE 1:2



PART #5 - CAP_BED-05
SCALE 1:2



PART #6 - CAP_BED_LEG_FRT
SCALE 1:2

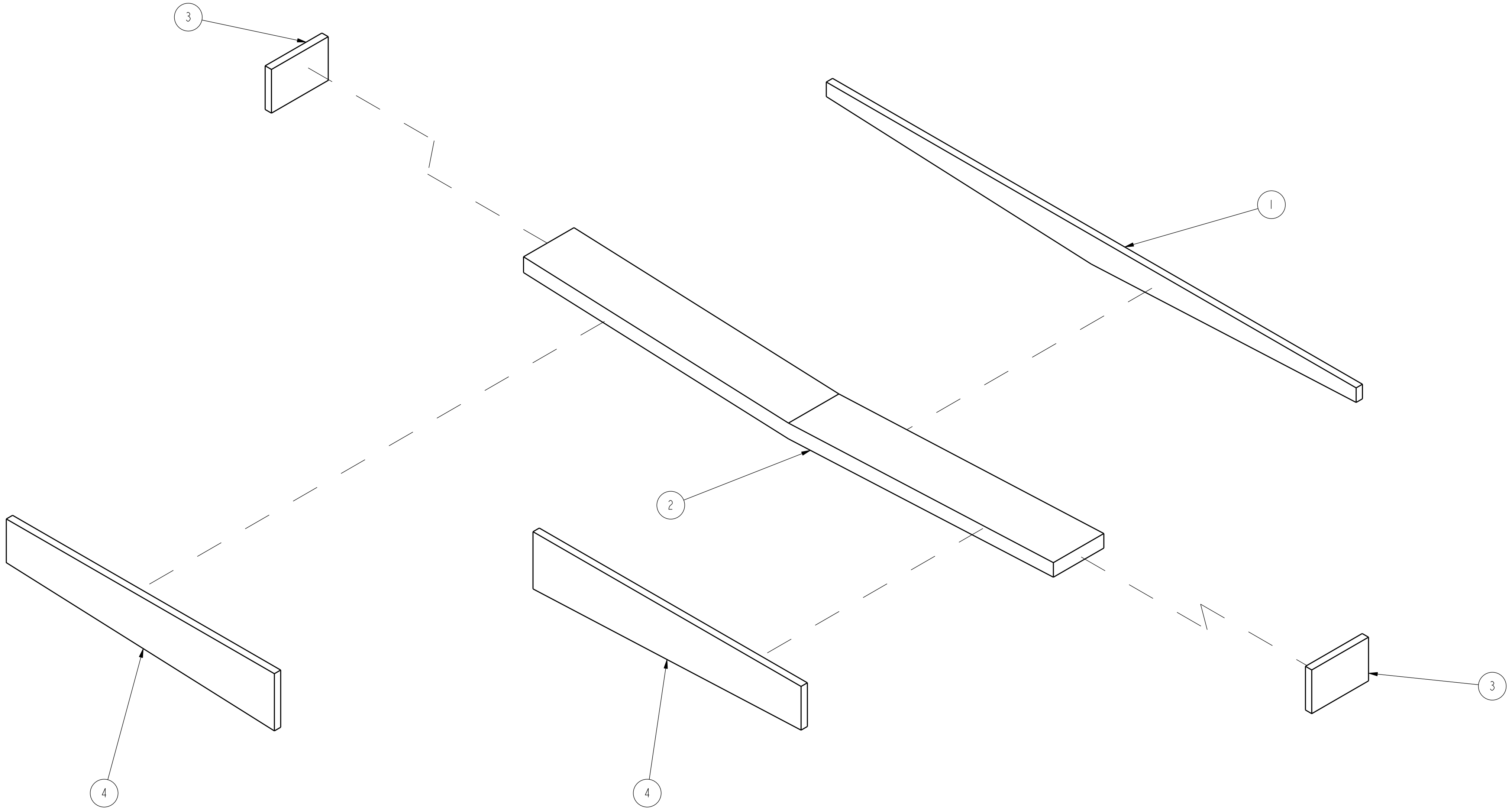


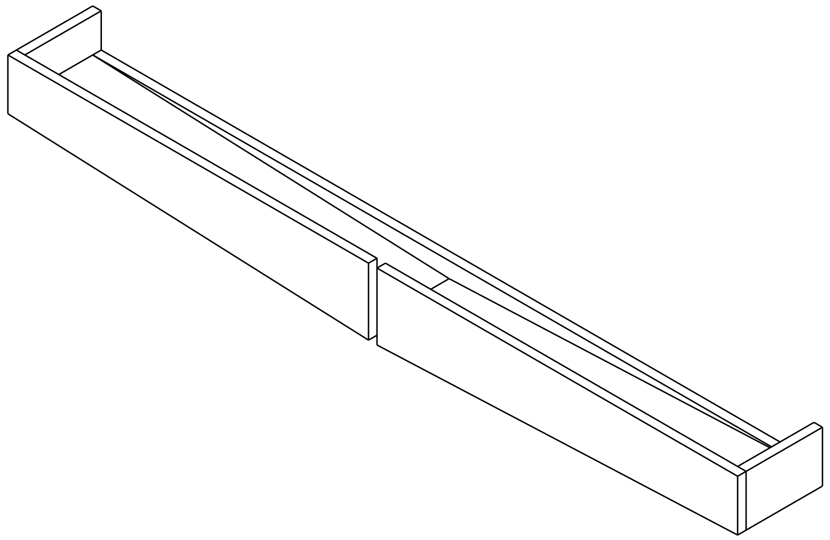
PART #7 - CAP_BED_LEG_RR
SCALE 1:2

SIZE	CAGE CODE	DWG NO: BASE_BED	REV
D	-	PART NO: -	1
SCALE: 3:10	CALC WT: 0.000	ACTUAL WT: -	SHEETS: 3 OF 3

INSPECTION POINT	TOTAL INSP. PTS : -

PART NO	SH	REV
-	1	1





NOTES:

1) USE APPROPRIATE ADHESIVE FOR ACRYLIC IN ASSEMBLY.

ALL ITEMS ABOVE ARE REFERENCE								
4	2		- (1/8 SHEET ACRYLIC-CLEAR [PLEXIGLASS] - / -)	CAP_BED_SPOUT-04	-		-	
3	2		- (1/8 SHEET ACRYLIC-CLEAR [PLEXIGLASS] - / -)	CAP_BED_SPOUT-03	-		-	
2	1		- (1/4 SHEET ACRYLIC-CLEAR [PLEXIGLASS] - / -)	CAP_BED_SPOUT-02	-		-	
1	1		- (1/8 SHEET ACRYLIC-CLEAR [PLEXIGLASS] - / -)	CAP_BED_SPOUT-01	-		-	
ITEM NO	QTY REQD	MATERIAL USAGE OR COMPONENT NUMBER	NOMENCLATURE OR DESCRIPTION	OWC NO	PART NO	CAGE CODE	CERT REQ	NOTE NO

DRAWING BILL OF MATERIALS

TOLERANCES
X.X ± 0.025 FRACTIONS ± 1/16
X.XX ± 0.010 ANGLES ± 0.5°
X.XXX ± 0.003

U.N.O. MACH. SURFACE FINISH 125
BREAK SHARP EDGES: 0.005 TO 0.010
FILLET RADII: 0.005 TO 0.010

THIRD ANGLE PROJECTION

DRAWING IN ACCORDANCE WITH ANSI Y14.5M

INCHES

DIMENSIONS ARE IN INCHES. DIMENSIONS SHOWN IN BRACKETS ARE MILLIMETERS.

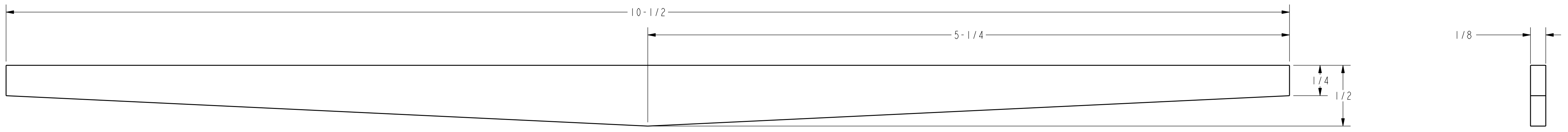
-	-
-	-
-	-
-	-
-	-
-	-
-	-
NEXT ASSEMBLY	USED ON
APPLICATION	

EXPERIMENTAL APPARATUS

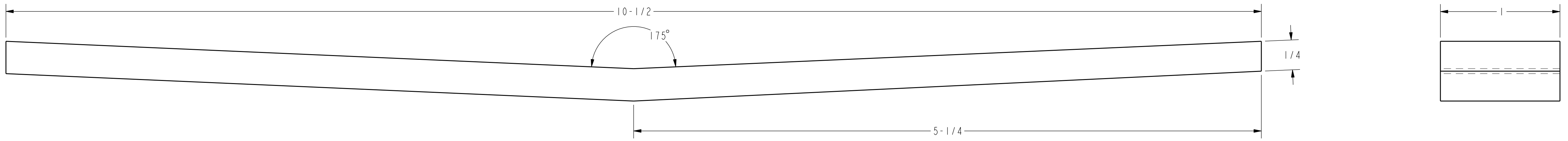
TEST BED

SPOUT/SAMPLE PORT

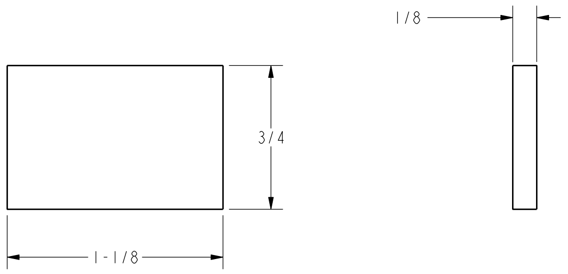
SIZE	CAGE CODE	PART NO:	REV
D		BASE_BED_SPOUT -	1
SCALE: 1:1	CALC WT: 0.000	ACTUAL WT: -	SHEET: 1 OF 2



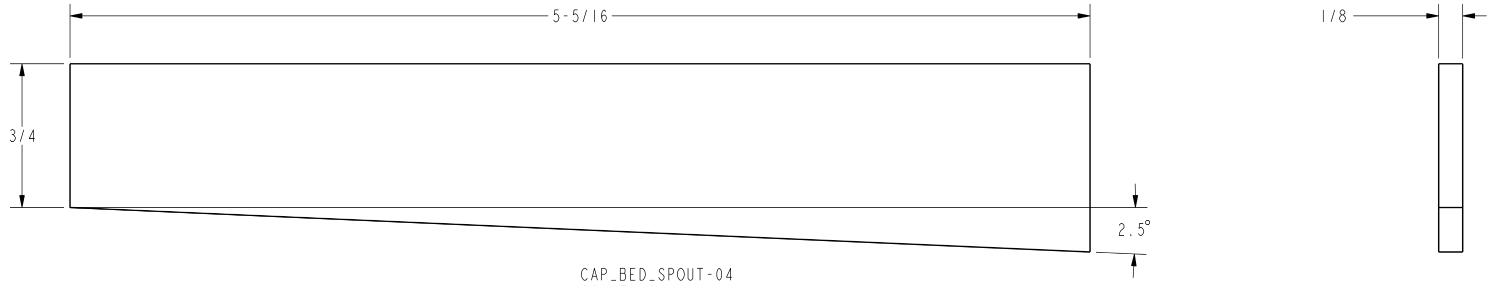
CAP_BED_SPOUT-01
SCALE 2:1



CAP_BED_SPOUT-02
SCALE 2:1



CAP_BED_SPOUT-03
SCALE 2:1

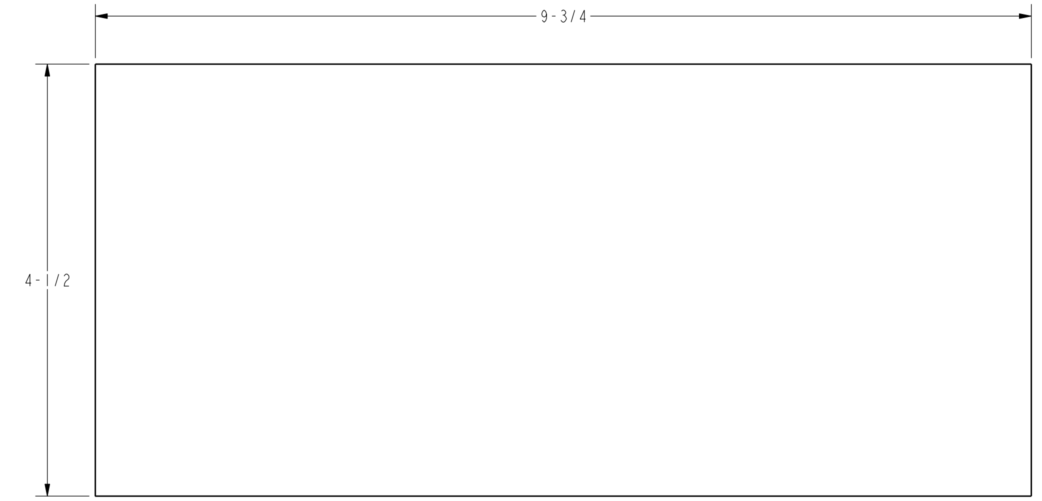


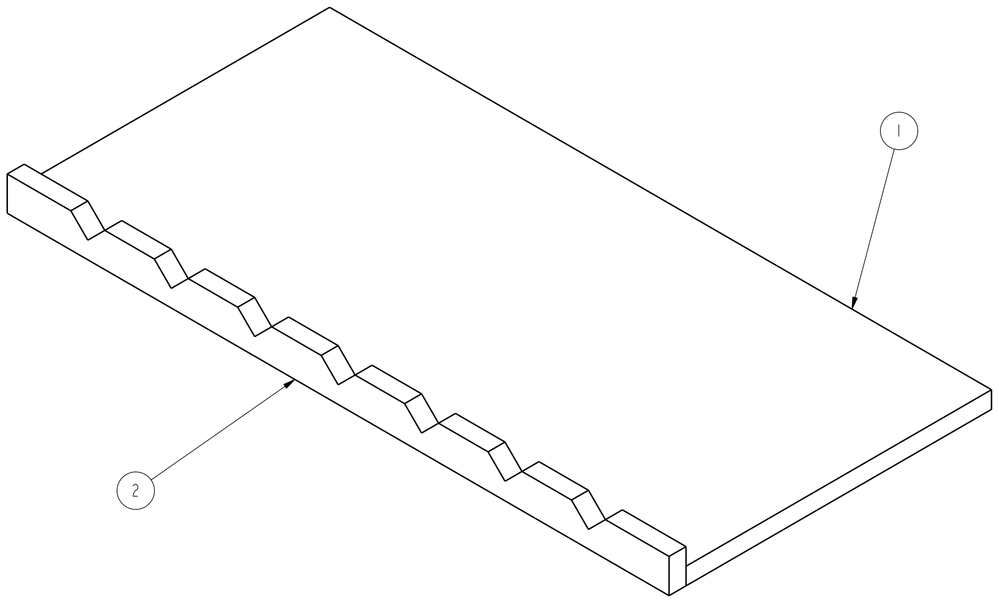
CAP_BED_SPOUT-04
SCALE 2:1

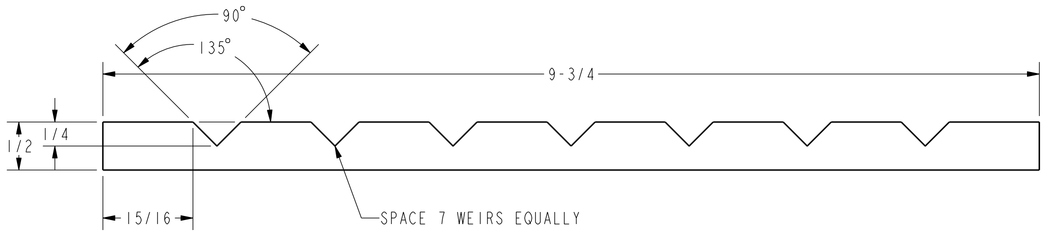
SIZE	CAGE CODE	DWG NO:	REV
D	-	BASE_BED_SPOUT	1
		PART NO: -	
SCALE: 1:1	CALC WT: 0.000	ACTUAL WT: -	SHEETS: 2 OF 2

INSPECTION POINT	TOTAL INSP. PTS : -

PART NO	SH	REV
-	1	1







NOTES:

1) USE APPROPRIATE ADHESIVE FOR ACRYLIC IN ASSEMBLY.

ITEM NO	QTY REQD	MATERIAL USAGE OR COMPONENT NUMBER	NOMENCLATURE OR DESCRIPTION	OWC NO	PART NO	CAGE CODE	CERT REQ	NOTE NO
ALL ITEMS ABOVE ARE REFERENCE								
2	1		- (3/8 SHEET ACRYLIC-CLEAR - / -)	7_WEIR_FLOW_DIST-02	-		-	
1	1		- (3/8 SHEET ACRYLIC-CLEAR - / -)	7_WEIR_FLOW_DIST-01	-		-	

DRAWING BILL OF MATERIALS

TOLERANCES
X.X ± 0.025 FRACTIONS ± 1/16
X.XX ± 0.010 ANGLES ± 0.5°
X.XXX ± 0.003

U.N.O. MACH. SURFACE FINISH 125
BREAK SHARP EDGES: 0.005 TO 0.010
FILLET RADII: 0.005 TO 0.010

THIRD ANGLE PROJECTION

DRAWING IN ACCORDANCE WITH ANSI Y14.5M

INCHES

DIMENSIONS ARE IN INCHES. DIMENSIONS SHOWN IN BRACKETS ARE MILLIMETERS.

NEXT ASSEMBLY	USED ON
-	-
-	-
-	-
-	-
-	-
-	-
-	-

APPLICATION

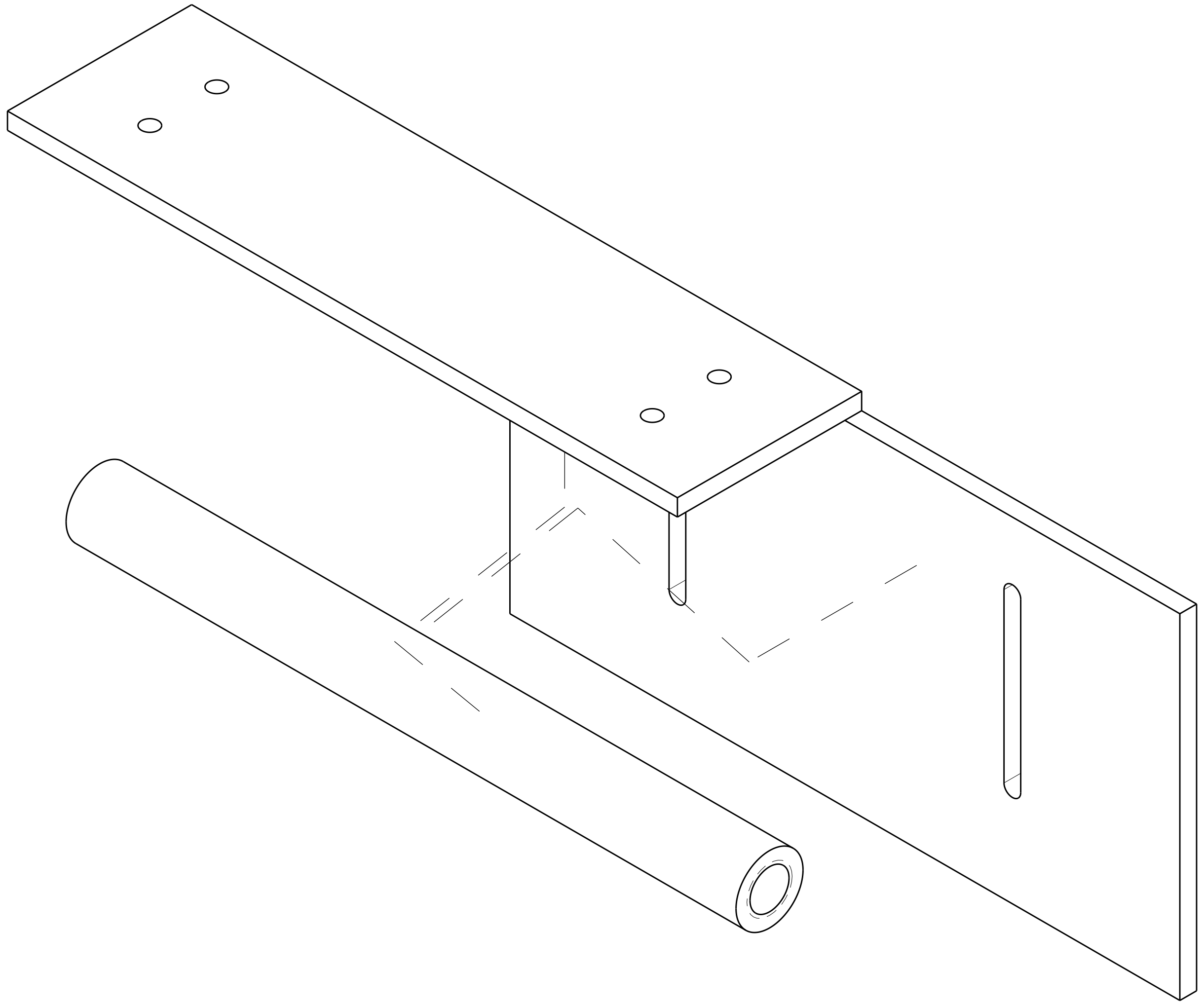
EXPERIMENTAL APPARATUS

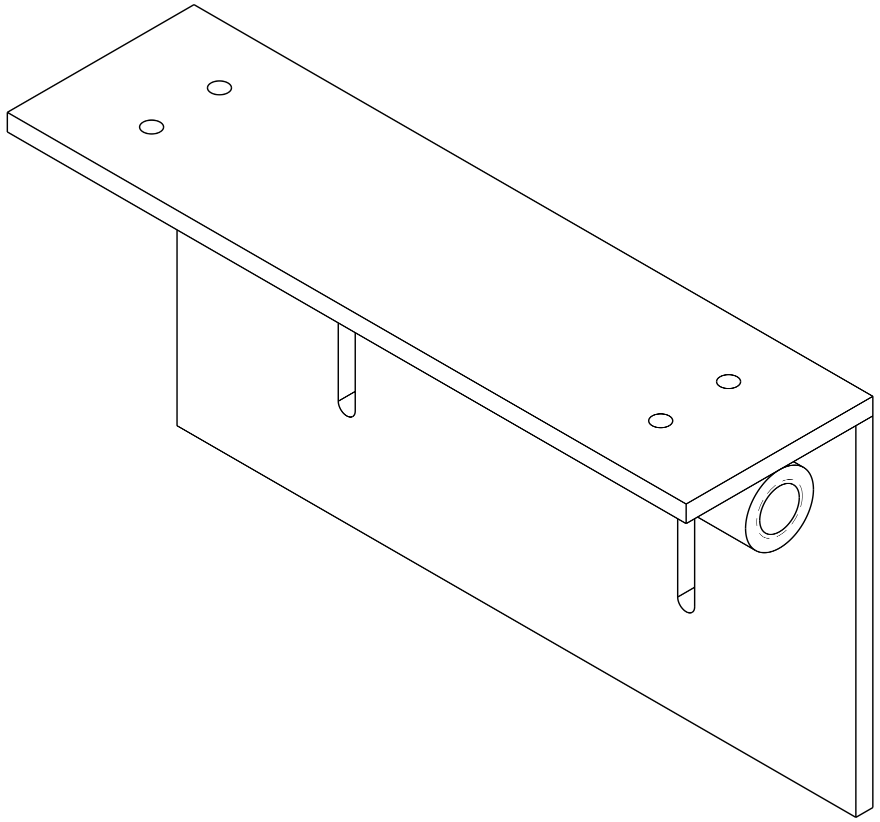
FLOW CHARACTERIZATION BED

7 WEIR FLOW DISTRIBUTOR

SIZE	CAGE CODE	PART NO:	REV
D		7_WEIR_FLOW_DIST -	1

SCALE: 1:1 | CALC WT: 0.000 | ACTUAL WT: - | SHEET: 1 OF 1





NOTE: ASSEMBLY NOT REQUIRED.

ALL ITEMS ABOVE ARE REFERENCE								
3	1	10 IN	- (1-1/4 ROUND BAR ACRYLIC CLEAR [PLEXIGLASS] - / -)	RAIN_SIM-03	-		-	
2	1	10 X 2-3/4 IN	- (1/4 PLATE ACRYLIC-CLEAR [PLEXIGLASS] - / -)	RAIN_SIM-02	-		-	
1	1	10 X 5 IN	- (1/4 PLATE ACRYLIC-CLEAR [PLEXIGLASS] - / -)	RAIN_SIM-01	-		-	
ITEM NO	QTY REQD	MATERIAL USAGE OR COMPONENT NUMBER	NOMENCLATURE OR DESCRIPTION	OWC NO	PART NO	CAGE CODE	CERT REQ	NOTE NO

DRAWING BILL OF MATERIALS

NOTES:

1) USE APPROPRIATE ADHESIVE FOR ACRYLIC IN ASSEMBLY.

TOLERANCES
X.X ± 0.025 FRACTIONS ± 1/16
X.XX ± 0.010 ANGLES ± 0.5°
X.XXX ± 0.003
U.N.O. MACH. SURFACE FINISH 125
BREAK SHARP EDGES: 0.005 TO 0.010
FILLET RADII: 0.005 TO 0.010
THIRD ANGLE PROJECTION
DRAWING IN ACCORDANCE WITH ANSI Y14.5M

INCHES
DIMENSIONS ARE IN INCHES. DIMENSIONS SHOWN IN BRACKETS ARE MILLIMETERS.

-	-
-	-
-	-
-	-
-	-
-	-
-	-
NEXT ASSEMBLY	USED ON
APPLICATION	

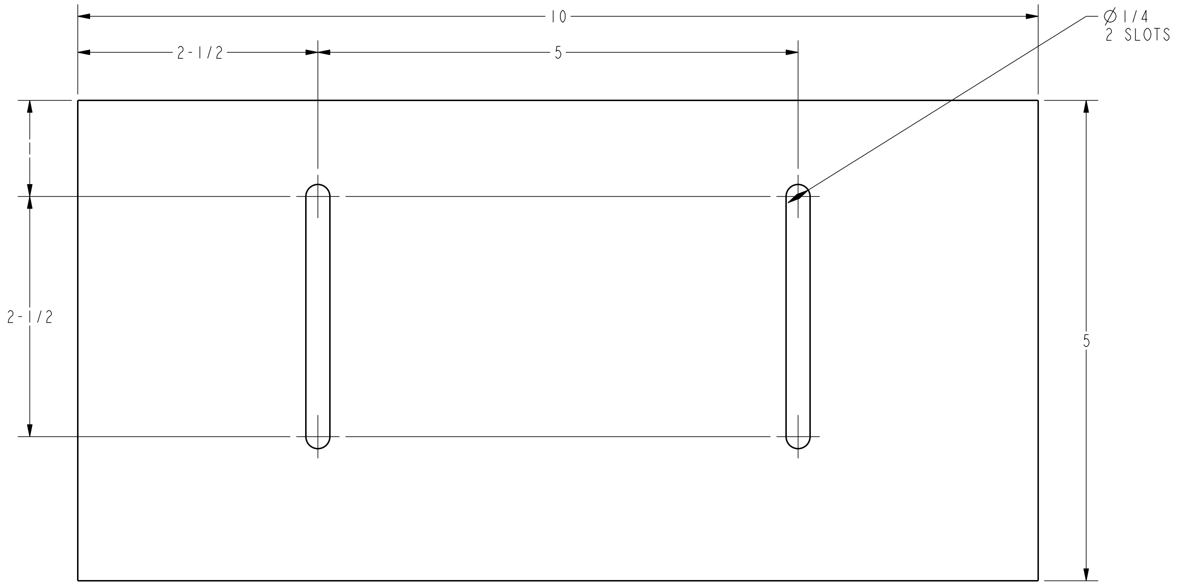
EXPERIMENTAL APPARATUS

ZINC CAPACITY BED

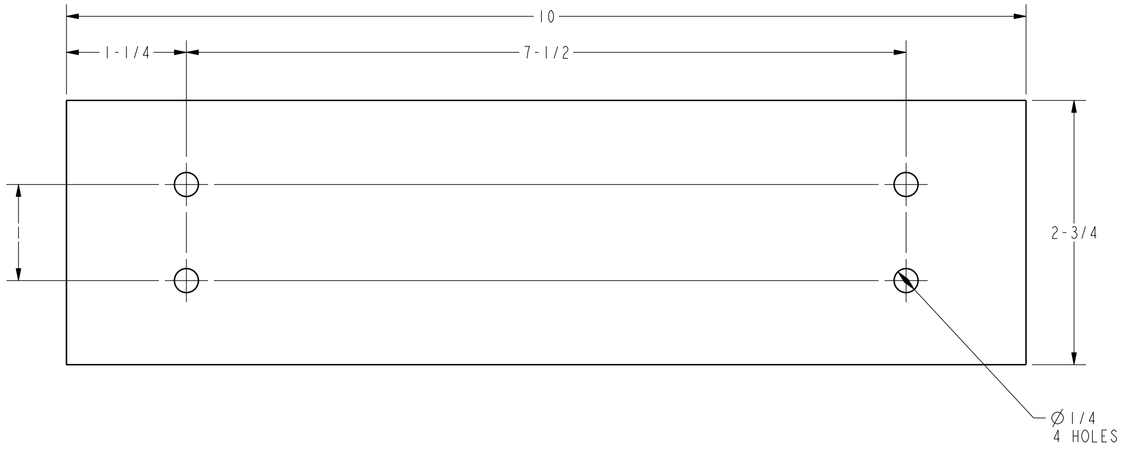
CYLINDRICAL FLOW DISTRIBUTOR

SIZE	CAGE CODE	PART NO:	REV
D		CYL_FLOW_DIST -	1
SCALE: 1:1	CALC WT: 0.000	ACTUAL WT: -	SHEET: 1 OF 2

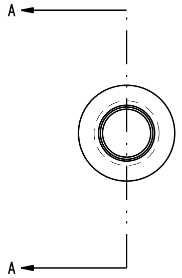
PART NO	SH	REV
-	2	1



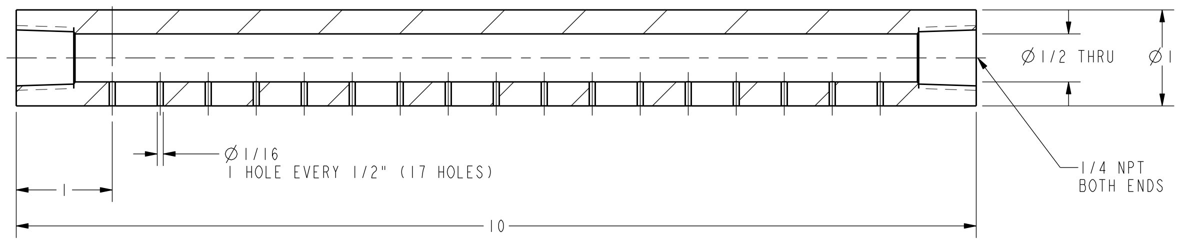
RAIN_SIM_01



RAIN_SIM_02



RAIN_SIM_03



SECTION A-A

SIZE	CAGE CODE	DWG NO: CYL_FLOW_DIST		REV
D		PART NO: -		1
SCALE: 1:1	CALC WT: 0.000	ACTUAL WT: -	SHEETS: 2 OF 2	

APPENDIX F – Quality Control Assurances for Bromide Tracer Experiments

Quality control tests were conducted on the flow distributor before its experimental use to verify that the weirs of the flow distributor contained comparable concentration gradients over time. Table 17 illustrates an example of one of several tests on the flow distributor weirs to ensure an even distribution of Br^- concentration.

Table 17- Flow Distributor: Weir Quality Control (70 ml/min)

Time (seconds)	[Br-] weir 2	[Br-] weir 6
Initial [DI water}	1.03E-04	7.86E-05
0-30	1.52E-01	1.60E-01
30-60	1.78E-01	1.83E-01
60-90	1.86E-01	1.94E-01
Initial [Br-]	1.94E-01	1.94E-01

A step-change bromide tracer event was conducted for each flow rate on the apparatus only which includes the tubing, flow distributor and tracer bed. The retention time of the apparatus was calculated for each flow rate and used to obtain a corrected retention time for the compost-only and vegetated compost configurations (see Table 18 below). The corresponding concentration-time graphs are presented in Figure 35, Figure 36, Figure 37 and Figure 38 below.

Table 18-Apparatus Only Retention Time

Storm Event	Apparatus Only Retention Time (seconds)
low flow (70 mL/min)	26.63
1-Year (264 mL/min)	13.91
2-Year (676 mL/min)	8.43
5-Year (900 mL/min)	6.92

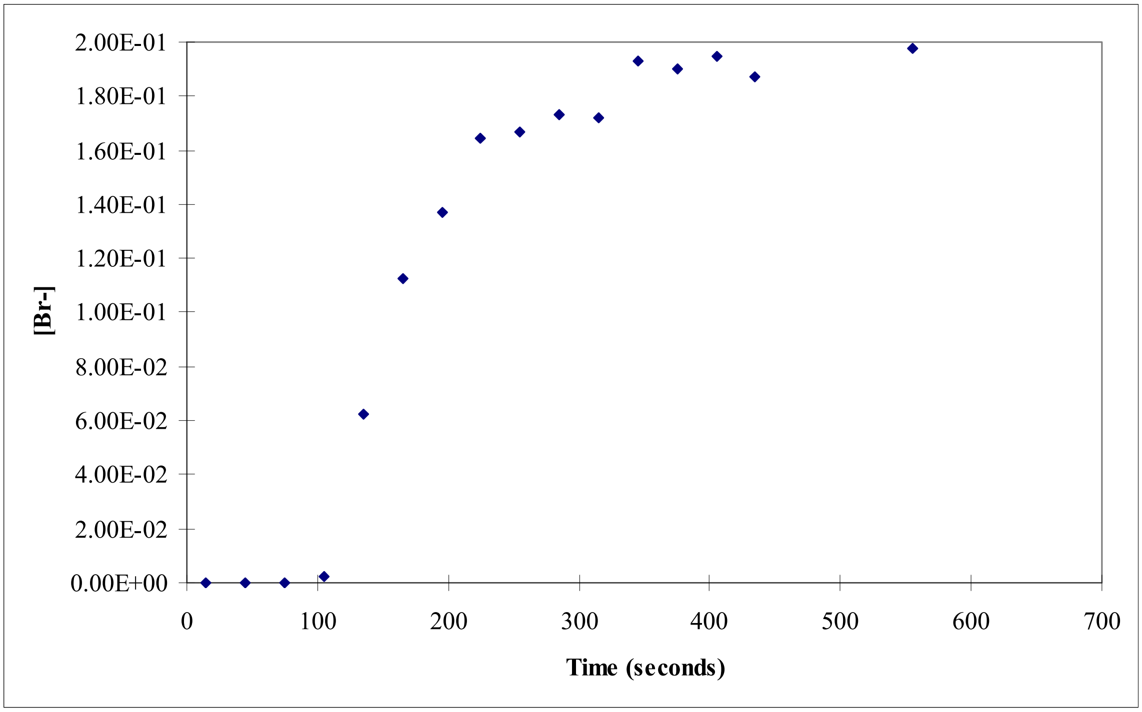


Figure 35- Low Flow (70 mL/min) Apparatus Only Tracer Run

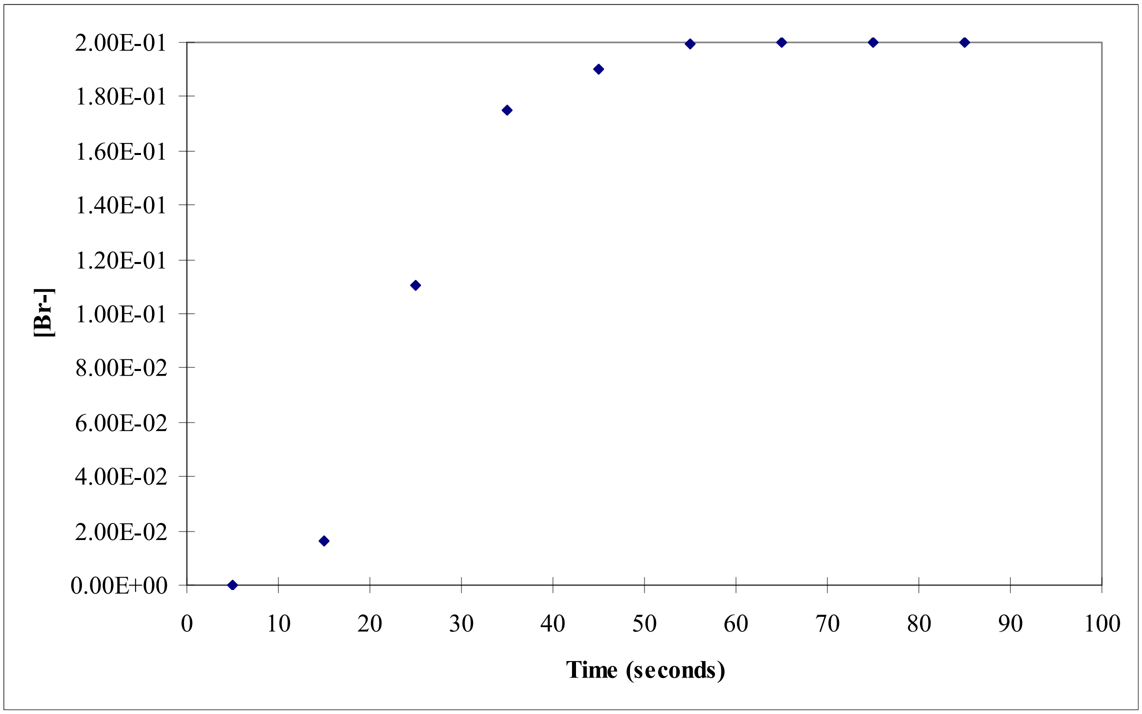


Figure 36- 1- Year (264 mL/min) Apparatus Only Tracer Run

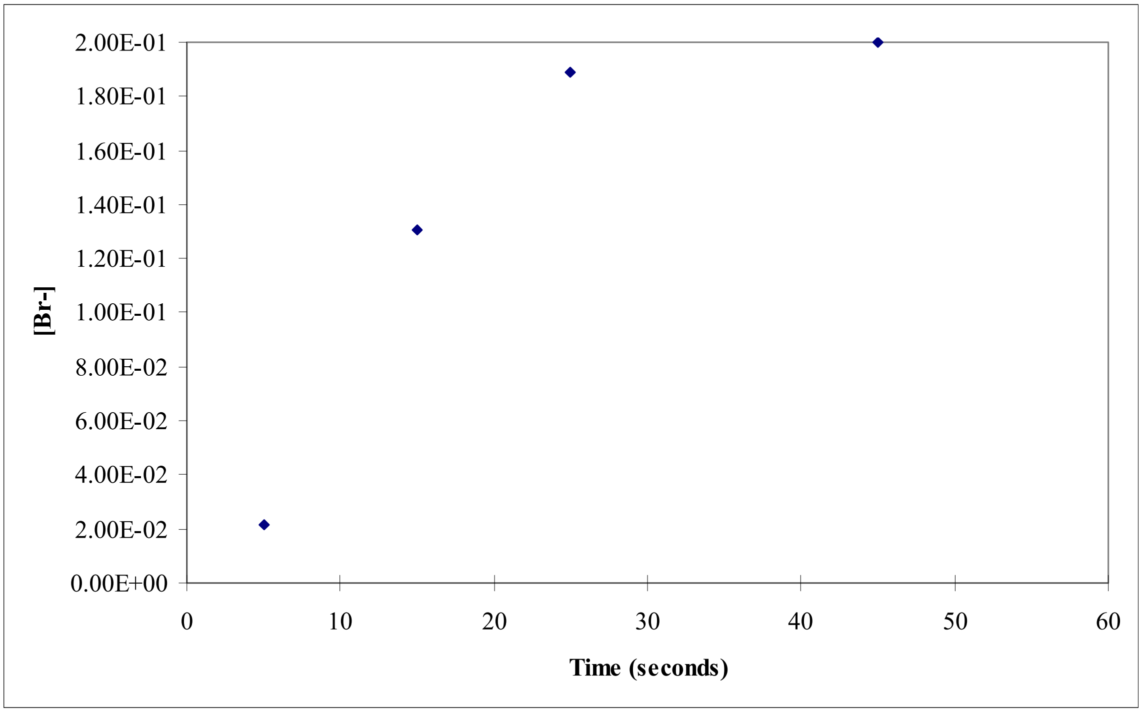


Figure 37- 2-Year (707 mL/min) Apparatus Only Tracer Run

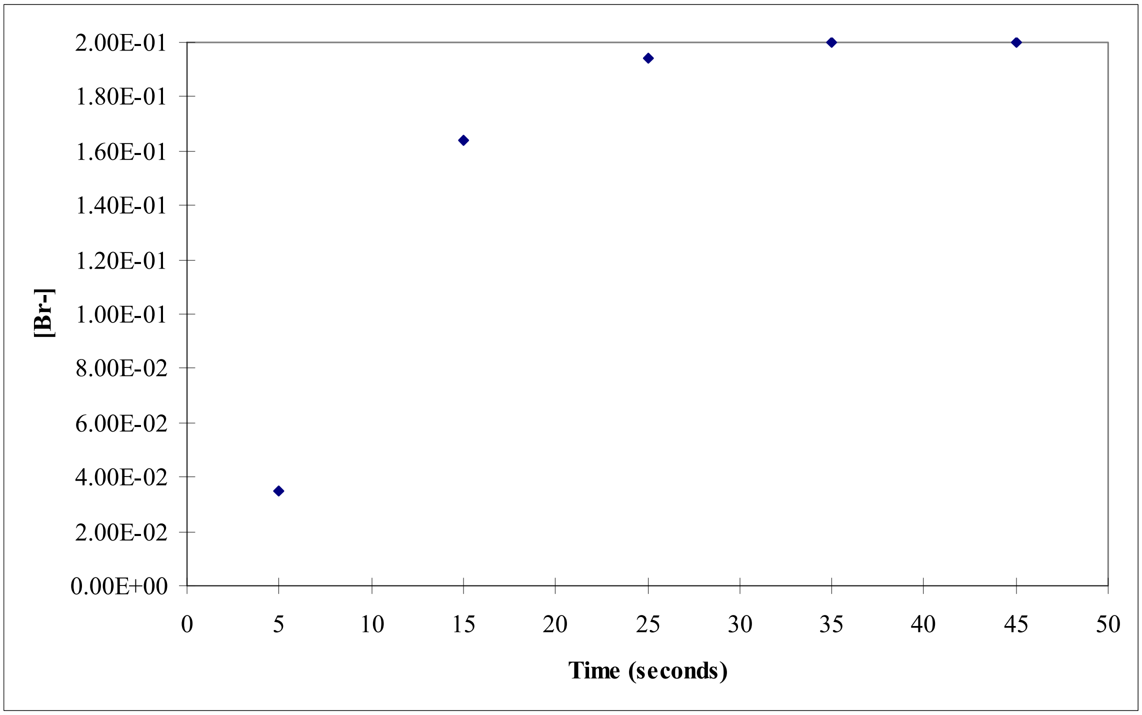


Figure 38- 5-Year (900 mL/min) Apparatus Only Tracer Run

Calibrating and Sample Measurement of Br$^-$

The millivolt potentials of Bromide standard solutions (100ppm) are measured with the bromide electrode. A calibration curve was then constructed on semi log graph paper Figure 39. Concentration was plotted on the log scale and mV on the linear scale. A slope of -59+-4mV (assuming solutions were measured between 20-30C) indicates correct electrode operation. For accurate measurement, the standards and the samples were analyzed at the same temperature. Ionic Strength Adjustor (ISA) was added to both standards and samples to 'swap out" standard and sample differences in ionic strength. 5.0 M Sodium Nitrate.

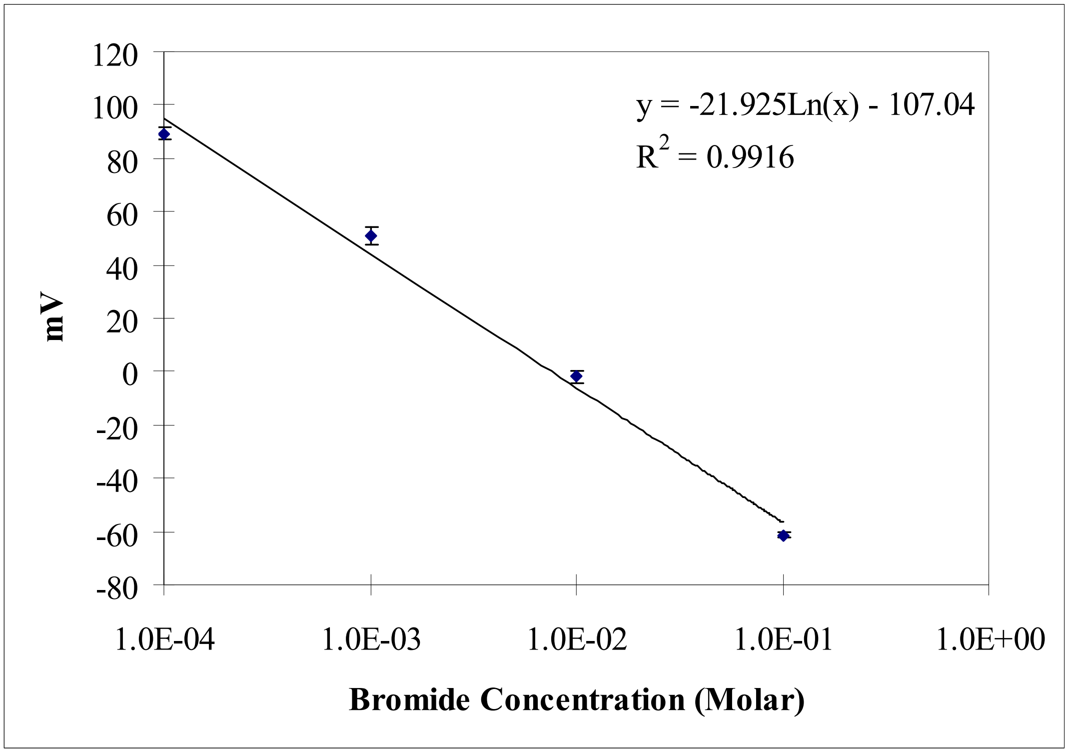


Figure 39- Accumet Bromide Probe Calibration Curve, n=3, error bars= one standard deviation

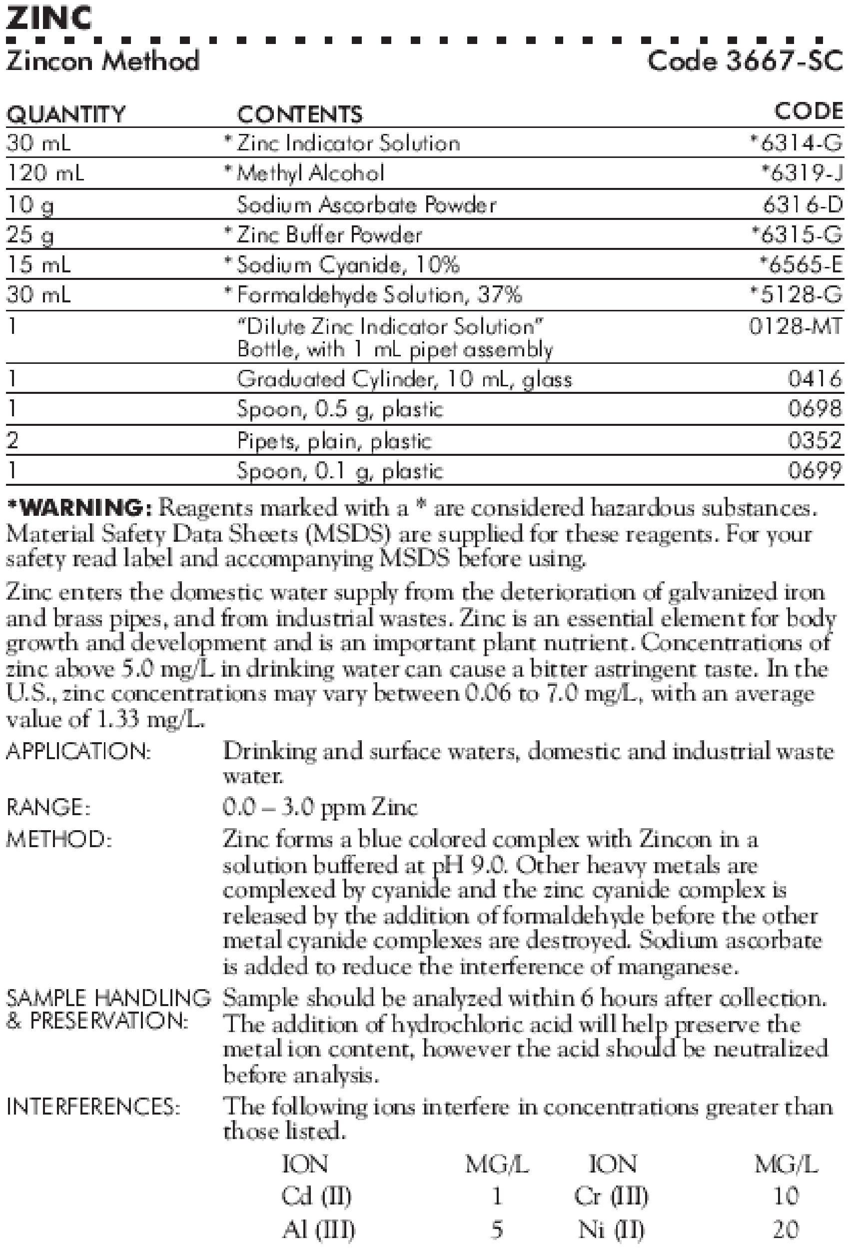
Bromide as a Conservative Tracer Verification

The procedure for verifying bromide as a conservative tracer in compost analysis was adopted from *Bromide as a conservative tracer for soil-water studies* (Levy and Chambers, 1987). Compost samples (2.5 g) were weighed and placed into Nalgene sample bottles which contained various concentrations of KBr solution (10^{-4}, 10^{-3}, 10^{-2}, 10^{-1}, 0.0 M). Samples were extracted every 2 hours and analyzed. The results presented in Table 19 conclude that bromide does not bind to compost over time and can be considered a conservative tracer for the purposes outlined in this paper.

Table 19- Bromide as a Conservative Tracer

	Time (hours)			
[Br-]	0	2	4	6
DI water	1.47E-06	1.68E-06	1.57E-06	1.80E-06
1.00E-04	1.96E-04	5.59E-05	4.80E-05	8.44E-05
1.00E-03	3.81E-04	4.35E-04	5.05E-04	6.25E-04
1.00E-02	6.29E-03	6.12E-03	6.01E-03	6.08E-03
1.00E-01	1.03E-01	1.02E-01	1.02E-01	1.04E-01

APPENDIX G - LaMotte Test Kit Procedure



ZINC

Zincon Method — **Code 3667-SC**

QUANTITY	CONTENTS	CODE
30 mL	*Zinc Indicator Solution	*6314-G
120 mL	*Methyl Alcohol	*6319-J
10 g	Sodium Ascorbate Powder	6316-D
25 g	*Zinc Buffer Powder	*6315-G
15 mL	*Sodium Cyanide, 10%	*6565-E
30 mL	*Formaldehyde Solution, 37%	*5128-G
1	"Dilute Zinc Indicator Solution" Bottle, with 1 mL pipet assembly	0128-MT
1	Graduated Cylinder, 10 mL, glass	0416
1	Spoon, 0.5 g, plastic	0698
2	Pipets, plain, plastic	0352
1	Spoon, 0.1 g, plastic	0699

***WARNING:** Reagents marked with a * are considered hazardous substances. Material Safety Data Sheets (MSDS) are supplied for these reagents. For your safety read label and accompanying MSDS before using.

Zinc enters the domestic water supply from the deterioration of galvanized iron and brass pipes, and from industrial wastes. Zinc is an essential element for body growth and development and is an important plant nutrient. Concentrations of zinc above 5.0 mg/L in drinking water can cause a bitter astringent taste. In the U.S., zinc concentrations may vary between 0.06 to 7.0 mg/L, with an average value of 1.33 mg/L.

APPLICATION:	Drinking and surface waters, domestic and industrial waste water.
RANGE:	0.0 – 3.0 ppm Zinc
METHOD:	Zinc forms a blue colored complex with Zincon in a solution buffered at pH 9.0. Other heavy metals are complexed by cyanide and the zinc cyanide complex is released by the addition of formaldehyde before the other metal cyanide complexes are destroyed. Sodium ascorbate is added to reduce the interference of manganese.
SAMPLE HANDLING & PRESERVATION:	Sample should be analyzed within 6 hours after collection. The addition of hydrochloric acid will help preserve the metal ion content, however the acid should be neutralized before analysis.
INTERFERENCES:	The following ions interfere in concentrations greater than those listed.

ION	MG/L	ION	MG/L
Cd (II)	1	Cr (III)	10
Al (III)	5	Ni (II)	20

Mn (II)	5	Co (II)	30
Fe (III)	7	CrO_4(II)	50
Fe (II)	9		

PROCEDURE

PREPARATION OF DILUTE ZINC INDICATOR SOLUTION

1. Use a pipet (0352) to measure exactly 5.0 mL of *Zinc Indicator Solution (6314) into 10 mL graduated cylinder (0416). The bottom of the curved surface (the meniscus) of liquid should be at 5.0 mL mark. Pour this into the bottle labeled “Dilute Zinc Indicator Solution”.
2. Use unrinsed graduated cylinder to add 10.0 mL and then 7.8 mL (total of 17.8 mL) of *Methyl Alcohol (6319) to bottle labeled “Dilute Zinc Indicator Solution”. Cap and mix ingredients in this bottle. Do not leave this bottle uncapped.

DETERMINATION OF ZINC

1. Press **USE** button to turn on colorimeter.
2. Scroll to and select ALL TESTS (or another sequence containing 42. ZINC) from TESTING MENU.
3. Scroll to and select 42. ZINC from menu.
4. Rinse a clean colorimeter tube (0967) with sample water. Fill to the 10 mL line with sample.
5. Insert tube into colorimeter chamber and select SCAN BLANK. (See Note)
6. Remove tube from colorimeter. Use 0.1 g spoon (0699) to add one measure of Sodium Ascorbate Powder (6316). Use 0.5 g spoon (0698) to add one measure of *Zinc Buffer Powder (6315). Cap and shake vigorously for 1 minute. Some undissolved buffer may remain in the bottom of the tube.
7. Add 3 drops of *Sodium Cyanide, 10% (6565). Cap and mix.
8. Use the 1 mL pipet assembly to add 1 mL of “Dilute Zinc Indicator Solution”. Cap and mix.
9. Use a second plain pipet (0352) to add 4 drops of *Formaldehyde Solution, 37% (5128). Cap and mix by inverting 15 times.
10. Insert tube into colorimeter chamber and select SCAN SAMPLE. Record result.
11. Press **OFF** button to exit to previous menu or make another menu selection.

NOTE: For best possible results, a reagent blank should be determined to account for any contribution to the test result by the reagent system. To determine the reagent blank, follow the above test procedure to scan a distilled or deionized water blank. Then follow the above procedure to perform the test on a distilled or deionized water sample. This test result is the reagent blank. Subtract the reagent blank from all subsequent test results of unknown samples. It is necessary to determine the reagent blank only when a new lot number of reagents are obtained.

APPENDIX H - Residence Time Distribution Considerations

1.0 Consideration of Method of Tracer Injection

Initially pulse injection was considered for the method of tracer injection. Pulse injection has several advantages including using less bromide tracer as well as generating C(t), E(t) and F(t) graphs with more information. However, after several trials, it was determined that the pulse input could not be replicated with enough accuracy to provide solid experimental results. Therefore, the method of tracer injection was chosen to be a step-change introduction. The step-change injection gave reliable, replicable results using 0.2 M of Br- tracer. Additional quality assurance, quality control data for bromide tracer protocol is located in Appendix E.

2.0 E(t) Exit Age Distribution Curves

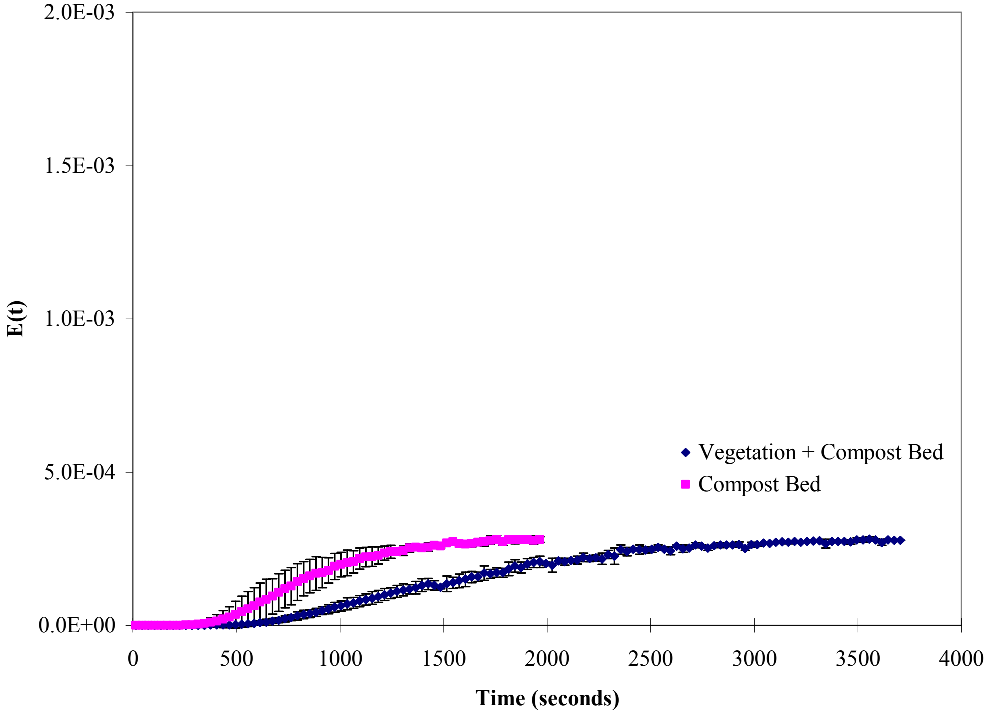


Figure 40- Low Flow, 70 ml/min Storm Event Exit Age Distribution Curve, n=3, error bars= one standard deviation

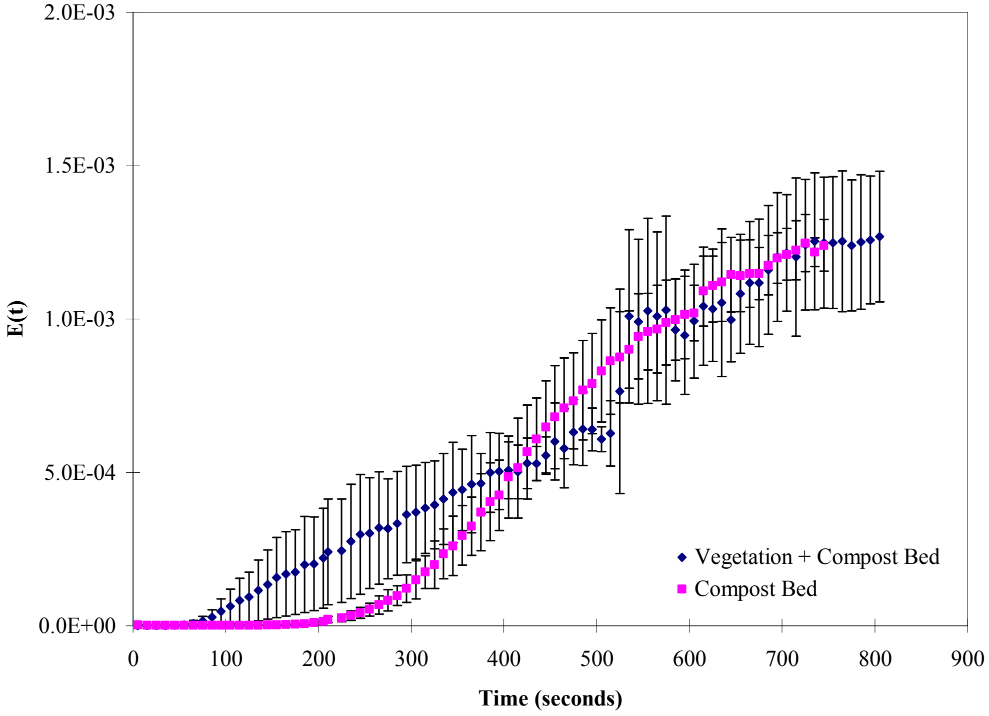


Figure 41- 1-year, 264 ml/min Storm Event Exit Age Distribution Curve, n=3, error bars= one standard deviation

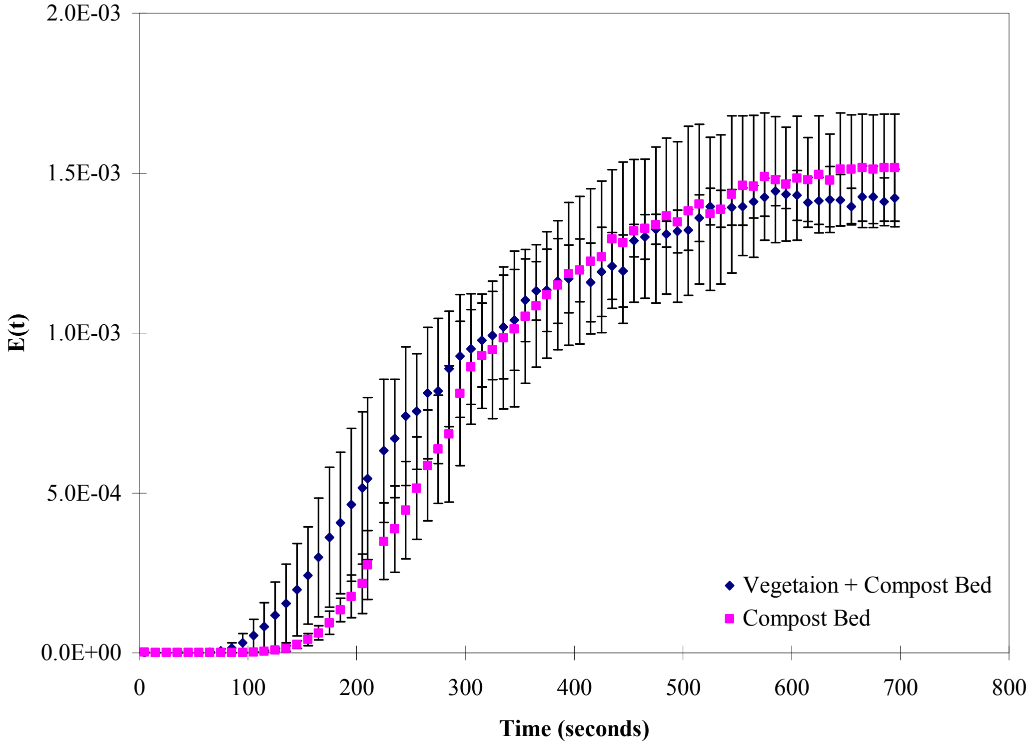


Figure 42- 2-year 600 ml/min Storm Event Exit Age Distribution Curve, n=3, error bars= one standard deviation

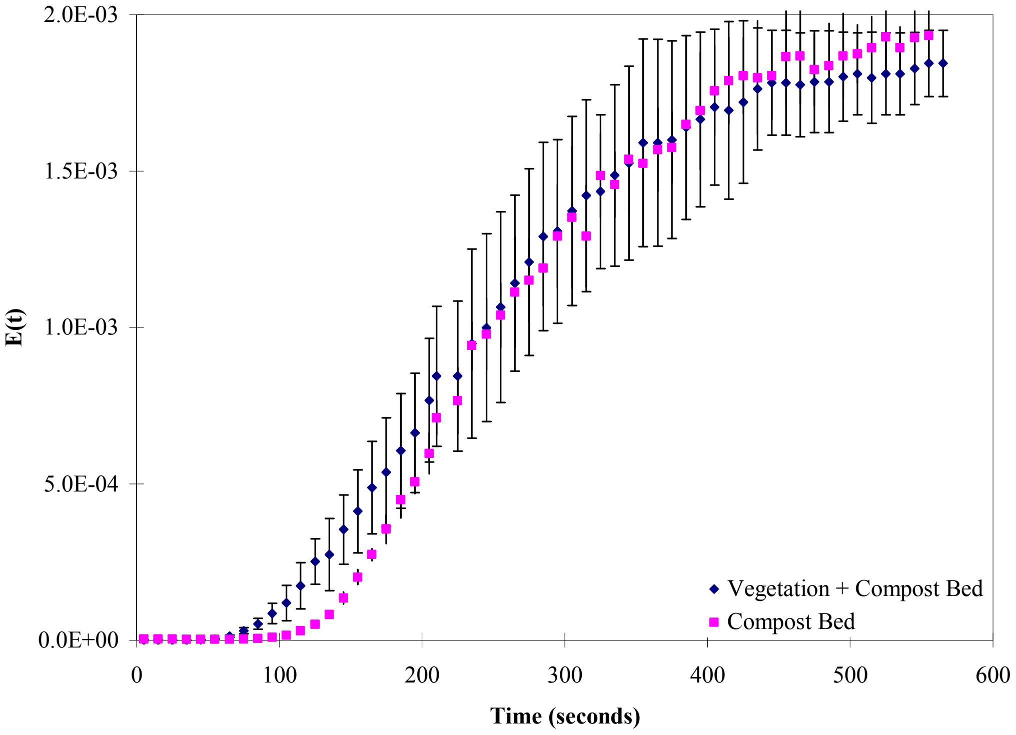


Figure 43- 5-year ml/min Storm Event Exit Age Distribution Curve, n=3, error bars= one standard deviation

3.0 F(t) Non-Dimensional Curves

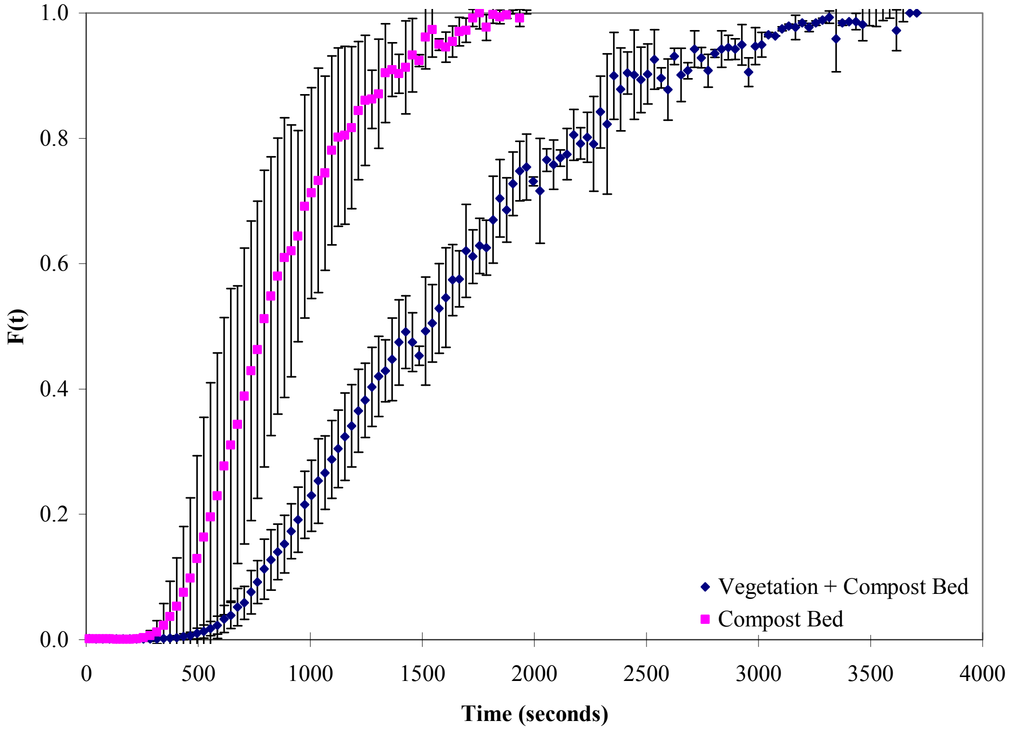


Figure 44- Low Flow, 70 ml/min Storm Event Non-Dimensional Curve, n=3, error bars= one standard deviation

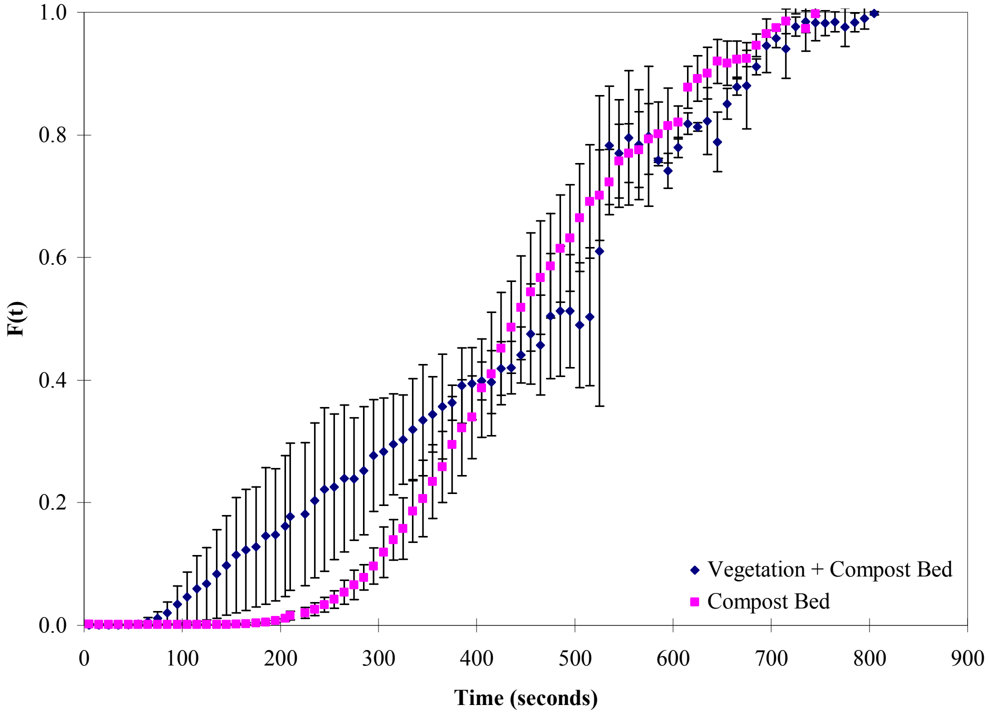


Figure 45- 1-year, 264 ml/min Storm Event Non-Dimensional Curve, n=3, error bars= one standard deviation

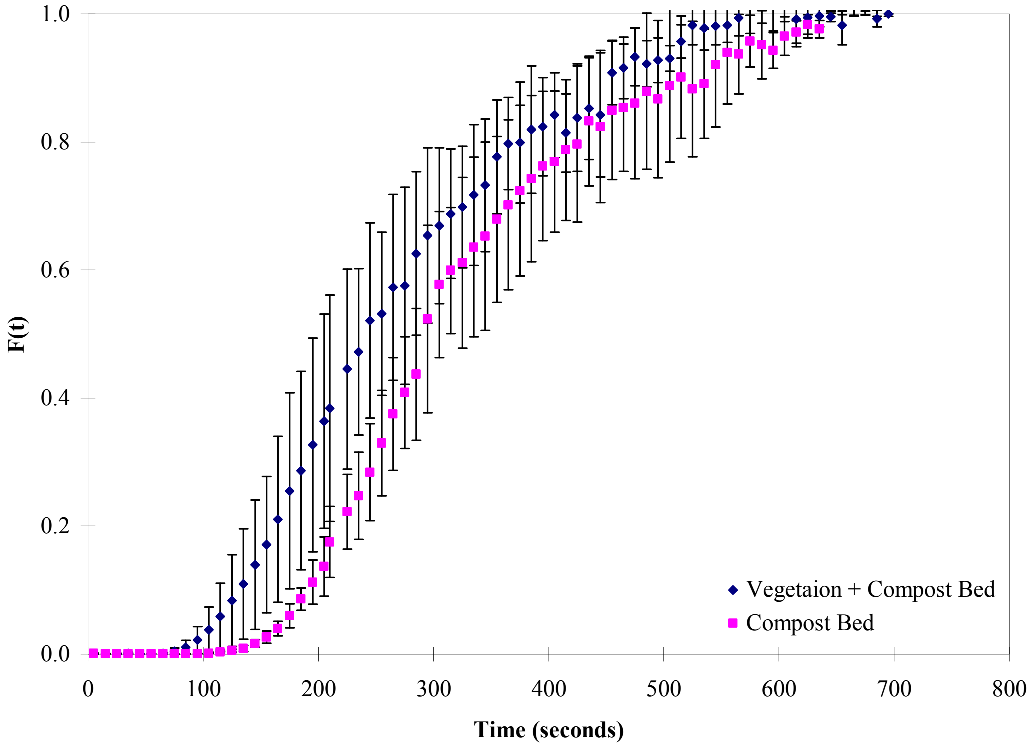


Figure 46- 2-year, 70 ml/min Storm Event Non-Dimensional Curve, n=3, error bars= one standard deviation

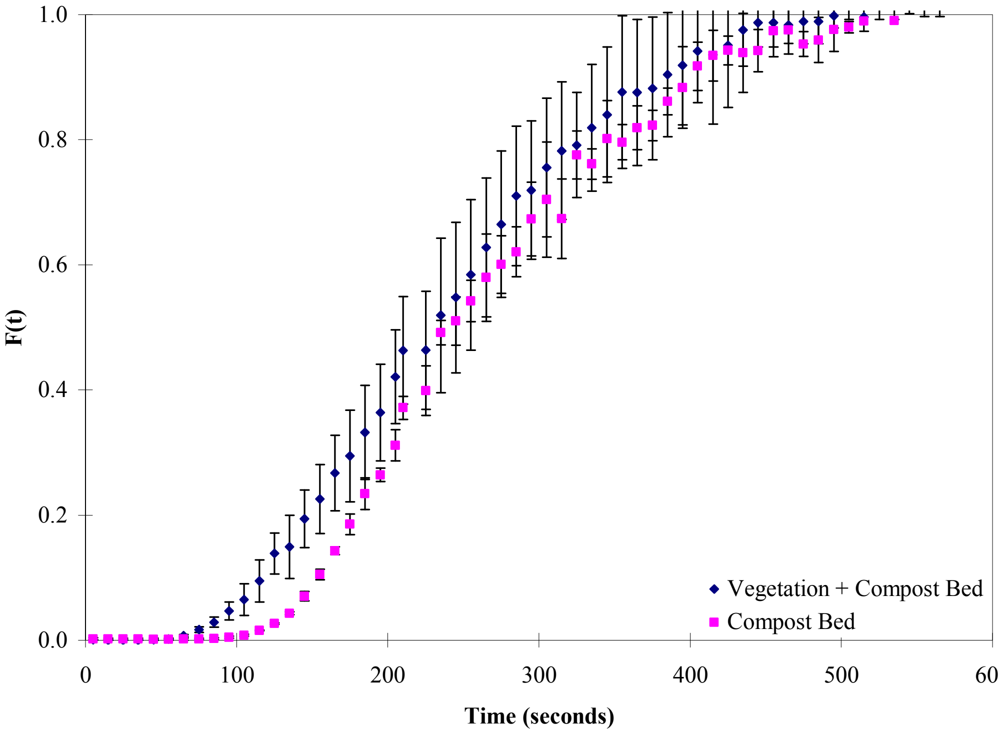


Figure 47- 5-year, 70 ml/min Storm Event Non-Dimensional Curve, n=3, error bars= one standard deviation

APPENDIX I- Raw Zinc Data

Table 20-Zinc Capacity Trial 1 Raw Data

		VEG 1->TOP			***N-VEG 1->BOTTOM***		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
26-Oct-08	0	0.43	71	6.6	0.56	49	6.68
	15	0	35	6.84	0.1	44	6.89
	45	0.08	20	6.74	0.05	24	6.74
	90	0.08	17	6.81	0.07	18	6.83
	120	0.04	13	6.79	0.03	16	6.75
	135	0.01	14	6.72	0.01	14	6.83
	180	0.08	12	6.64	0.01	14	6.69
	225	0.08	10	6.73	0.11	13	6.72
	270	0.01	12	6.68	0.6	13	6.67
27-Oct-08	**270**	**0.27**	**127**	**6.95**	**0.16**	**145**	**6.88**
	285	0.04	26	7.14	0.26	27	6.83
	360	0.04	11	6.82	0.07	13	6.8
	420	0.04	9	6.44	0.08	11	6.54
28-Oct-08	420	0.26	115	6.52	0.18	167	6.66
	525	0.05	10	7.05	0.12	12	6.85
	540	0.05	10	6.29	0.42	10	6.57
	570	0.46	8	6.25	0.08	9	6.53
	585	0.05	8	6.4	0.16	9	6.54
	600	0.14	7	6.45	0.07	9	6.53
	615	0.15	8	6.64	0.38	9	6.48
29-Oct-08	615	0.22	169	6.58	0.22	179	6.68
	630	0.15	20	6.87	0.17	20	6.73
	705	0.15	8	6.5	0.14	8	6.69
	810	0.17	10	6.95	0.21	8	6.79
6-Nov-08	810	0.2	118	6.67	0.21	123	6.89
	960	0.14	9	8.02	0.14	10	8.03
	1125	0.15	9	7.32	0.29	10	7.25
7-Nov-08	1125	0.3	109	6.8	0.48	115	6.9
	1200	0.29	9	6.62	0.29	9	6.89
	1320	0.28	8	6.6	0.13	8	6.69
	1410	0.24	8	6.64	0.4	9	6.5

		VEG 1->TOP			*N-VEG 1->BOTTOM*		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
8-Nov-08	1410	0.4	96	6.75	0.46	93	6.92
	1530	0.19	8	6.73	0.24	10	6.84
	1605	0.25	8	6.88	0.35	8	6.96
9-Nov-08	1605	0.33	67	6.81	0.47	98	6.95
	1650	0.19	11	6.58	0.17	11	6.64
	1695	0.22	8	6.55	0.8	7	6.6
	1710	--	--	--	0.18	11	6.59
	1770	0.16	7	6.53	0.2	9	6.58
10-Nov-08	1770	0.29	63	6.77	0.27	79	6.87
	1875	0.2	7	7.11	0.2	9	7.04
	1995	0.26	7	6.85	0.29	9	6.96
11-Nov-08	1995	0.35	64	6.84	0.51	71	6.91
	2175	0.32	5	6.33	0.31	8	6.26
	2295	0.41	5	6.39	0.39	9	6.32
12-Nov-08	2295	0.44	48	6.36	0.43	69	6.68
	2385	0.41	6	6.51	0.33	9	6.7
	2535	0.34	6	6.57	0.26	8	6.51
13-Nov-08	2535	0.46	46	6.62	0.42	62	6.81
	2715	0.35	7	6.66	0.35	10	6.78
17-Nov-08	2715	0.55	100	6.48	0.65	132	6.48
	2800	0.4	5	6.62	0.32	9	6.58
	2980	0.46	6	6.56	0.47	11	6.62
18-Nov-08	2980	0.52	67	6.78	0.52	71	6.71
	3240	0.62	8	6.61	0.5	10	6.81
19-Nov-08	3240	0.51	51	6.05	0.48	61	6.09
	3465	0.52	8	6.19	0.48	9	6.13
20-Nov-08	3840	0.6	8	6.94	0.58	7	6.85
21-Nov-08	3840	0.58	40	6.44	0.51	57	6.56
	4150	0.68	7	6.55	0.63	10	6.58
23-Nov-08	4150	0.6	43	6.63	0.53	51	6.81
	4675	0.74	7	6.27	--	--	--
	4735	1.1	6	6.42	0.68	8	6.37

		VEG 1->TOP			***N-VEG 1->BOTTOM***		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
24-Nov-08	4735	0.95	31	6.46	0.85	35	6.51
	5180	0.65	5	6.48	0.5	7	6.44
25-Nov-08	5180	0.57	56	6.25	0.57	67	6.13
	5645	0.95	5	6.38	0.9	10	6.43
26-Nov-08	5645	1.4	36	6.43	1	42	6.51
	6045	1	6	6.42	1.13	6	6.39
27-Nov-08	6045	1.4	37	6.3	1.15	51	6.4
	6345	1.3	6	6.33	0.8	7	6.35
	6450	0.9	6	6.38	1.8	7	6.43
28-Nov-08	6450	1.55	34	6.84	1.4	44	6.62
	7185	1.1	6	5.82	1.75	41	6.72
	7485	--	5	6.13	1.15	--	--
1-Dec-08	7485	2.05	56	6.1	1.65	68	6.13
	8240	1.7	6	6.11	1.7	9	6.25
2-Dec-08	8240	1.55	36	6.05	1.55	41	6.13
	8515	1	5	6.17	0.8	7	6.22
4-Dec-08	8620	1.8	35	6.57	1.88	42	6.64
	8910	0.88	5	6.36	1.8	7	6.51
5-Dec-08	9030	1.9	31	5.82	1.8	37	5.95
	9435	1.9	4	6.02	1.8	6	5.97
6-Dec-08	9435	2.1	29	5.96	2	36	6.02
	9845	1.85	5	6.62	1.8	7	6.35
7-Dec-08	9845	2.7	22	5.97	2.7	29	5.92
	9935	1.9	6	5.93	1.5	8	5.92
	10005	1.8	5	6	1.7	7	5.95
8-Dec-08	10245	1.9	23	5.94	1.9	7	6.49
	10545	1.8	7	6.36	2.2	29	6.3
	10800	1.5	6	6.02	1.4	6	6.04
10-Dec-08	10800	2.1	23	5.99	2	27	5.98
	10995	1.6	6	5.97	1.5	7	5.99
11-Dec-08	10995	2.2	22	6.01	2	26	6.04
	11325	2.1	4	6.08	1.5	4	6.05
12-Dec-08	11325	2.4	21	6.02	2.2	24	6.07
	11670	2.5	5	5.97	2.5	6	6.01
13-Dec-08	11670	3	19	5.95	2.5	22	5.98

		VEG 1->TOP			*N-VEG 1->BOTTOM*		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
14-Dec-08	11970	2.3	27	6.12	2.5	34	6.15
	12240	2.2	8	6.36	2.6	9	6.43
15-Dec-08	12240	3	19	6.51	2.8	21	6.32
	12900	2.5	6	6.22	2.2	7	6.23
12-Jan-09	12900	3	20	6.15	2.8	22	6.12
	13110	2.6	6	5.99	2.5	7	6.06
16-Jan-09	13110	3	18	5.91	2.9	19	5.97
	13340	2.7	6	5.78	2.6	8	5.9
18-Jan-09	13340	2.9	21	5.96	2.8	23	5.99
	13700	2.8	6	5.76	2.9	7	5.87
19-Jan-09	13700	3	19	5.9	2.9	22	5.88
	14140	2.8	7	5.78	3	8	5.82
20-Jan-09	14140	2.9	17	5.78	2.9	19	5.81
	14520	3	5	5.83	3	7	5.87
21-Jan-09	14520	2.9	17	5.87	3	18	5.96
	14880	3	5	5.89	3	7	5.85

Table 21- Zinc Capacity Trial 2 Raw Data

		VEG 2->TOP			*N-VEG 2->BOTTOM*		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
Feb 11-2009	30	0.26	6	6.72	0.04	7	6.88
	390	0.41	6	6.49	0.02	7	6.68
Feb 12-2009	390	0.52	52	6.42	0.15	19	6.54
	720	0.005	6	6.95	0.02	10	7.07
Feb 13-2009	720	0.027	53	6.66	0.3	88	6.6
	1125	0.14	7	6.84	0.07	8	6.75
Feb 15-2009	1125	0.31	48	6.5	0.25	78	6.46
	1245	0.19	7	6.63	0.12	11	6.59
Feb 16-2009	1245	0.21	47	6.29	0.13	68	6.4
	1365	0.18	6	6.5	0.13	10	6.51
	1625	0.23	6	6.49	0.19	7	6.49
Feb 17-2009	1625	0.26	42	6.47	0.21	48	6.42
	1775	0.26	4	6.5	0.16	6	6.51
	2030	0.24	4	6.21	0.29	8	5.94
Feb 18-2009	2030	0.4	38	6.38	0.37	58	6.24
	2195	0.31	6	6.44	0.24	7	6.46
Feb 19-2009	2195	0.46	34	6.41	0.46	38	6.4
	2360	0.3	4	6.53	0.31	8	6.54
	2645	0.29	7	6.54	0.39	7	6.55
Feb 20-2009	2645	0.47	32	6.46	0.58	31	6.46
	2730	0.5	2	6.44	0.6	4	6.3
	3370	0.29	2	6.53	0.42	5	6.51
	3790	--	10	--	--	--	--

		VEG 2->TOP			***N-VEG 2->BOTTOM***		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
Feb 21-2009	3790	0.44	10	6.57	0.46	13	6.53
	4195	0.4	3	6.63	0.43	5	6.56
	4435	0.4	3	6.59	0.42	3	6.65
Feb 22-2009	4435	1.3	28	6.55	1.2	29	6.54
	4635	0.4	3	6.64	0.48	4	6.62
	4955	0.47	2	6.61	0.45	3	6.59
Feb 23-2009	4955	0.52	29	6.56	0.58	30	6.49
	5325	1.2	3	6.64	1.4	6	6.61
Feb 24-2009	5325	1.4	30	6.58	2	30	6.5
	5615	1.3	2	6.65	1	5	6.63
	5905	1.9	3	6.55	1.9	3	6.58
Feb 25-2009	5905	1.3	24	6.53	1.3	25	6.5
	6225	1.7	2	6.52	1.7	3	6.58
	6450	1.4	2	6.56	1.5	4	6.59
Feb 26-2009	6450	1.3	24	6.57	1.8	26	6.52
	6870	1.5	3	6.55	1.6	4	6.51
Feb 27-2009	6870	1.8	25	6.54	2.1	31	6.48
	7290	1.8	3	6.49	1.9	4	6.44
Mar 5-2009	7290	1.9	53	6.26	2.1	58	6.41
	7575	1.7	4	6.38	1.3	36	6.35
Mar 6-2009	7575	2.05	32	6.35	1.65	38	6.35
	7930	1.7	6	6.4	1.7	9	6.39
	8320	1.55	36	6.37	1.55	41	6.31
Mar 7-2009	8320	1.8	25	6.29	1.8	26	6.37
	8650	1.9	4	6.52	1.7	4	6.66
	9040	1.8	5	6.62	1.8	3	6.62
Mar 8-2009	9040	2.2	21	6.71	2.4	23	6.44
	9385	1.8	3	6.42	1.9	4	6.73

		VEG 2->TOP			***N-VEG 2->BOTTOM***		
Date	**Time**	**[Zn] ppm**	**turbidity (FTU)**	**pH**	**[Zn] ppm**	**turbidity (FTU)**	**pH**
Mar 9-2009	9385	1.9	20	6.37	2	22	6.54
	9775	1.8	4	6.38	1.6	4	6.63
Mar 10-2009	9775	2.1	19	6.22	2	22	6.5
	10135	1.9	6	6.31	2.2	5	6.45
Mar 11-2009	10135	2	21	6.28	2.4	20	6.39
	10515	1.8	4	6.31	1.6	6	6.37
Mar 12-2009	10515	2.4	19	6.38	2.5	18	6.27
	11265	2.7	3	6.26	2.6	6	6.31
Mar 16-2009	11265	2.7	20	6.11	2.8	21	6.24
	11655	2.2	4	6.13	2.2	6	6.05
Apr 9-2009	11655	2.6	26	6.25	2.8	24	6.07
	11895	2.4	3	6.13	2.5	4	6.01
Apr 17-2009	11895	2.8	24	6.27	2.9	21	5.98
	12165	2.5	4	6.22	2.6	4	6.15
Apr 20-2009	12165	3	21	6.08	3.1	23	6.43
	12435	2.7	4	6.17	2.8	3	6.29
Apr 22-2009	12435	2.9	22	6.09	2.9	18	6.23
	12795	2.7	4	5.95	2.9	4	5.99
Apr 29-2009	12795	2.9	19	6.03	2.8	21	6.04
	13125	2.8	4	5.92	2.9	3	6.09
Apr 30-2009	13125	3.2	18	6.01	3	17	6.12
	13515	3	4	5.85	3	4	5.98
May 2-2009	13515	3.1	17	5.9	3.2	16	5.92
	13785	3	4	5.94	3.1	3	5.96

APPENDIX J- Quality Control Assurances for Zinc Sorption Experiments

Zinc Stock Preparation Calculations:

5.0g/L Zinc Solution = 5.0 g of Zinc

98% $ZnCl_2$ (company name)

Molar Weight Zn= 65.41 g/mol

Molar Weight Cl= 35.45 g/mol

Molar Weight $ZnCl_2$ = 135.31 g/mol

$$\frac{65.41\text{ g Zinc}}{135.31\text{ g }ZnCl_2} = 0.4834\ \frac{\text{g Zinc}}{\text{g }ZnCl_2}$$

$C_1V_1=C_2V_2$

$C_1=ZnCl_2$

$V_1=x$

$C_2=Zn$

$V_2=5g$

X=10.34 g $ZnCl_2$

Since it is 98% $ZnCl_2$

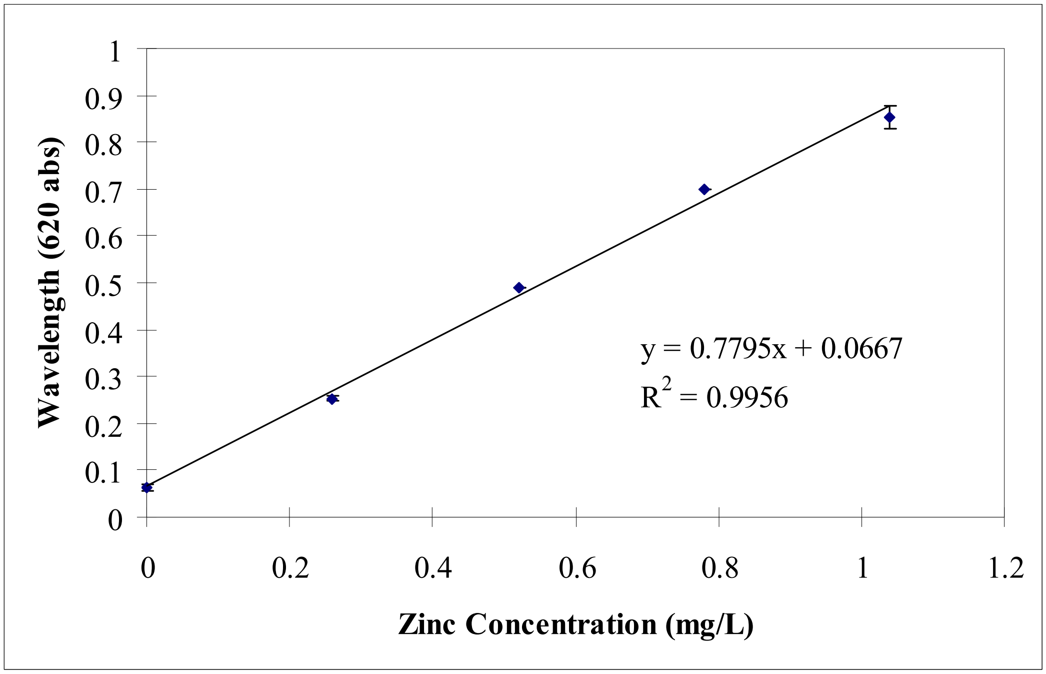


Figure 48-Colorimeter Zinc Calibration Curve

Table 22-Zinc Capacity Trial 1- Quality Control

incoming water temperature 9.9°C and incoming turbidity 4 FTU

	Stock Tank		Veg Bed	No Veg Bed	Colorimeter Standard
Date	[Zn] ppm	pH	Flow (mL/Min)	Flow (mL/Min)	[Zn]= 2.0 ppm
26-Oct-08	2.90	5.70	265.0	265.0	2.0
27-Oct-08	3.00	5.62	270.0	265.0	
28-Oct-08	3.00	5.69	268.0	270.0	
29-Oct-08	3.00	5.59	270.0	265.0	1.9
06-Nov-08	2.80	5.55	265.0	262.0	
07-Nov-08	2.80	5.78	270.0	270.0	
08-Nov-08	2.90	5.65	265.0	265.0	2.0
09-Nov-08	2.90	5.52	265.0	265.0	
10-Nov-08	2.80	5.57	270.0	270.0	
11-Nov-08	2.80	5.56	272.0	270.0	
12-Nov-08	2.70	5.62	270.0	270.0	2.1
13-Nov-08	3.00	5.64	265.0	265.0	
17-Nov-08	3.10	5.59	265.0	265.0	
18-Nov-08	3.00	5.49	265.0	265.0	2.0
19-Nov-08	2.90	5.52	265.0	265.0	
20-Nov-08	3.00	5.57	255.0	258.0	
21-Nov-08	2.70	5.54	260.0	258.0	1.8
23-Nov-08	2.70	5.55	240.0	235.0	
24-Nov-08	3.00	5.61	268.0	268.0	
25-Nov-08	2.90	5.68	264.0	270.0	2.0
26-Nov-08	2.90	5.73	270.0	265.0	
27-Nov-08	2.70	5.58	265.0	265.0	
28-Nov-08	3.00	5.54	265.0	265.0	2.2
01-Dec-08	3.10	5.59	265.0	265.0	
02-Dec-08	2.80	5.63	265.0	265.0	
04-Dec-08	2.90	5.68	265.0	265.0	2.0
05-Dec-08	2.70	5.74	265.0	265.0	
06-Dec-08	3.00	5.55	265.0	265.0	
07-Dec-08	2.90	5.59	265.0	265.0	
08-Dec-08	2.80	5.62	265.0	265.0	2.1
10-Dec-08	--	5.69	265.0	265.0	--
11-Dec-08	--	5.58	265.0	265.0	--
12-Dec-08	--	5.68	265.0	265.0	--
13-Dec-08	--	5.52	265.0	265.0	--
14-Dec-08	--	5.59	265.0	265.0	--
15-Dec-08	--	5.61	265.0	265.0	--
12-Jan-09	2.90	5.67	265.0	265.0	2.0
16-Jan-09	2.80	5.60	265.0	265.0	
18-Jan-09	3.00	5.58	265.0	265.0	
19-Jan-09	3.05	5.62	265.0	265.0	2.0
20-Jan-09	3.11	5.56	265.0	265.0	
21-Jan-09	3.00	5.61	265.0	265.0	

-- zinc stock samples or colorimeter standard not conducted because of low amounts of zinc buffer

Table 23- Zinc Capacity Trial 2- Quality Control

incoming water temperature 9.9°C and incoming turbidity 4 FTU

	Stock Tank		*Veg Bed*	*N-Veg Bed*	*Colorimeter Standard*
Date	**[Zn] ppm**	**pH**	**Flow (mL/Min)**	**Flow (mL/Min)**	**[Zn]= 2.0 ppm**
Feb 11-2009	2.90	5.70	265.0	265.0	2.0
Feb 12-2009	2.70	5.74	270.0	265.0	
Feb 13-2009	3.00	5.69	268.0	270.0	
Feb 15-2009	2.70	5.59	270.0	265.0	2.1
Feb 16-2009	2.60	5.55	265.0	262.0	
Feb 17-2009	2.80	5.78	270.0	270.0	
Feb 18-2009	2.70	5.57	270.0	270.0	
Feb 19-2009	2.80	5.56	272.0	270.0	
Feb 20-2009	2.70	5.62	270.0	270.0	
Feb 21-2009	2.90	5.64	265.0	265.0	2.0
Feb 22-2009	2.90	5.59	265.0	265.0	
Feb 23-2009	2.90	5.62	265.0	265.0	
Feb 24-2009	3.00	5.50	265.0	265.0	
Feb 25-2009	3.10	5.55	255.0	258.0	2.0
Feb 26-2009	2.90	5.42	260.0	258.0	
Feb 27-2009	3.00	5.58	240.0	235.0	
Mar 5-2009	3.20	5.54	268.0	268.0	
Mar 6-2009	2.90	5.59	264.0	270.0	
Mar 7-2009	2.80	5.53	270.0	265.0	
Mar 8-2009	3.10	5.56	265.0	265.0	
Mar 9-2009	3.00	5.49	265.0	265.0	
Mar 10-2009	3.10	5.55	265.0	265.0	2.0
Mar 11-2009	2.90	5.59	265.0	265.0	
Mar 12-2009	3.10	5.62	265.0	265.0	
Mar 16-2009	3.30	5.69	265.0	265.0	
Apr 9-2009	3.60	5.71	265.0	265.0	
Apr 17-2009	3.20	5.67	265.0	265.0	
Apr 20-2009	3.10	5.59	265.0	265.0	
Apr 22-2009	3.00	5.63	265.0	265.0	2.1
Apr 29-2009	3.20	5.68	265.0	265.0	
Apr 30-2009	3.10	5.74	265.0	265.0	
May 2-2009	3.30	5.65	265.0	265.0	

APPENDIX K- Statistical Analysis of Results

Statistical Analysis of pH Trial 1:

PH of the vegetative compost bed effluent changed over time in trial 1 (F=164, p<<0.0005, df=1,109)

SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.775077
R Square	0.600745
Adjusted R	0.597082
Standard E	0.248413
Observatio	111

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	10.12079691	10.12079691	164.008	1.81328E-23
Residual	109	6.726278762	0.061708979		
Total	110	16.84707568			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	6.766562	0.035654123	189.7834485	3E-139	6.695897067	6.837227687	6.695897067	6.837227687
X Variable	-6.31E-05	4.92712E-06	-12.80658018	1.8E-23	-7.28649E-05	-5.33342E-05	-7.2865E-05	-5.3334E-05

PH of the compost bed effluent changed over time in trial 1 (F=200, p<<0.0005, df=1, 109)

SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.804908
R Square	0.647877
Adjusted R	0.644647
Standard E	0.226895
Observatio	111

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	10.32466877	10.32466877	200.551	1.85868E-26
Residual	109	5.611478974	0.051481458		
Total	110	15.93614775			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	6.81477	0.032565743	209.261911	8E-144	6.750225428	6.879313913	6.750225428	6.879313913
X Variable	-6.37E-05	4.50033E-06	-14.16161042	1.9E-26	-7.26514E-05	-5.48124E-05	-7.2651E-05	-5.4812E-05

Statistical Analysis of pH Trial 2:

There is a positive relationship between change in pH of the veg bed effluent and time (F=88.21, p<<0.0005, df=1,72)
SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.742021436
R Square	0.550595812
Adjusted R S	0.544354087
Standard Err	0.153527252
Observations	74

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	2.07921422	2.07921422	88.2121	3.90497E-14
Residual	72	1.697084429	0.023570617		
Total	73	3.776298649			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	6.6697558	0.033802638	197.3146525	4E-100	6.602371473	6.737140127	6.602371473	6.737140127
X Variable 1	-3.98995E-05	4.24818E-06	-9.392131016	3.9E-14	-4.83681E-05	-3.14309E-05	-4.8368E-05	-3.1431E-05

There is a positive relationship between change in pH of the compost bed effluent and time (F=48.5, p<<0.0005, df=1,72)
SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.634423939
R Square	0.402493735
Adjusted R S	0.394195037
Standard Err	0.179513387
Observations	74

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	1.562941914	1.562941914	48.5008	1.28644E-09
Residual	72	2.320204032	0.032225056		
Total	73	3.883145946			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	6.650789598	0.039524097	168.2717644	3.3E-95	6.571999751	6.729579445	6.571999751	6.729579445
X Variable 1	-3.45931E-05	4.96723E-06	-6.964253621	1.3E-09	-4.4495E-05	-2.46911E-05	-4.4495E-05	-2.4691E-05

Trial 1- Initial Turbidity

t-Test: Paired Two Sample for Means
(tstat=4.37, p=0.00008, df=41)

	Variable 1	*Variable 2*
Mean	49.30952381	57.26190476
Variance	1337.389663	1816.198026
Observations	42	42
Pearson Correlation	0.967238013	
Hypothesized Mean	0	
df	41	
t Stat	-4.376369266	
P(T<=t) one-tail	4.05718E-05	
t Critical one-tail	1.682878003	
P(T<=t) two-tail	8.11436E-05	
t Critical two-tail	2.019540948	

Conclusion= significantly different

Trial 1- Runtime Turbidity

t-Test: Paired Two Sample for Means
(tstat=8.6, p<<<0.0001, df=68)

	Variable 1	*Variable 2*
Mean	8.44928	10.08695652
Variance	25.7511	32.43350384
Observations	69	69
Pearson Correlation	0.96342	
Hypothesized Mean D	0	
df	68	
t Stat	-8.60514	
P(T<=t) one-tail	8.7E-13	
t Critical one-tail	1.66757	
P(T<=t) two-tail	1.7E-12	
t Critical two-tail	1.99547	

Conclusion= significantly different

Trial 2: Midrun Turbidity

t-Test: Paired Two Sample for Means

(tstat=5.44, p<<.0001, df=41)

	Variable 1	*Variable 2*
Mean	4.09524	5.428571429
Variance	2.28339	4.933797909
Observations	42	42
Pearson Correlation	0.69968	
Hypothesized Mean Diffe	0	
df	41	
t Stat	-5.44296	
P(T<=t) one-tail	1.3E-06	
t Critical one-tail	1.68288	
P(T<=t) two-tail	2.7E-06	
t Critical two-tail	2.01954	

Conclusion= significantly different

Trial 2: Initial Turbidity

t-Test: Paired Two Sample for Means

(tstat=1.70, p=0.099, df=31)

	Variable 1	*Variable 2*
Mean	28.09375	31.4688
Variance	148.6683468	365.16
Observations	32	32
Pearson Correlation	0.831463586	
Hypothesized Mean Difference	0	
df	31	
t Stat	-1.698341847	
P(T<=t) one-tail	0.049730278	
t Critical one-tail	1.695518742	
P(T<=t) two-tail	0.099460557	
t Critical two-tail	2.039513438	

Conclusion: not significantly different

Trial 2: Initial Turbidity with first 2 and last 10 readings not included

t-Test: Paired Two Sample for Means

(tstat=1.70, p=0.099, df=31)

	Variable 1	*Variable 2*
Mean	31.7	39.1
Variance	140.7473684	421.147
Observations	20	20
Pearson Correlation	0.931851988	
Hypothesized Mean Difference	0	
df	19	
t Stat	-3.182282566	
P(T<=t) one-tail	0.002452551	
t Critical one-tail	1.729132792	
P(T<=t) two-tail	0.004905102	
t Critical two-tail	2.09302405	

Conclusion= significantly different

APPENDIX L- Photographs of Pooling in Compost Bed

Figure 49- Low Flow Rate- 70 ml/min, Compost Only Bed

Figure 50- 1 Year Storm Event- 264 ml/min, Compost Only Bed

Figure 51- 2 Year Storm Event- 676 ml/min, Compost Only Bed

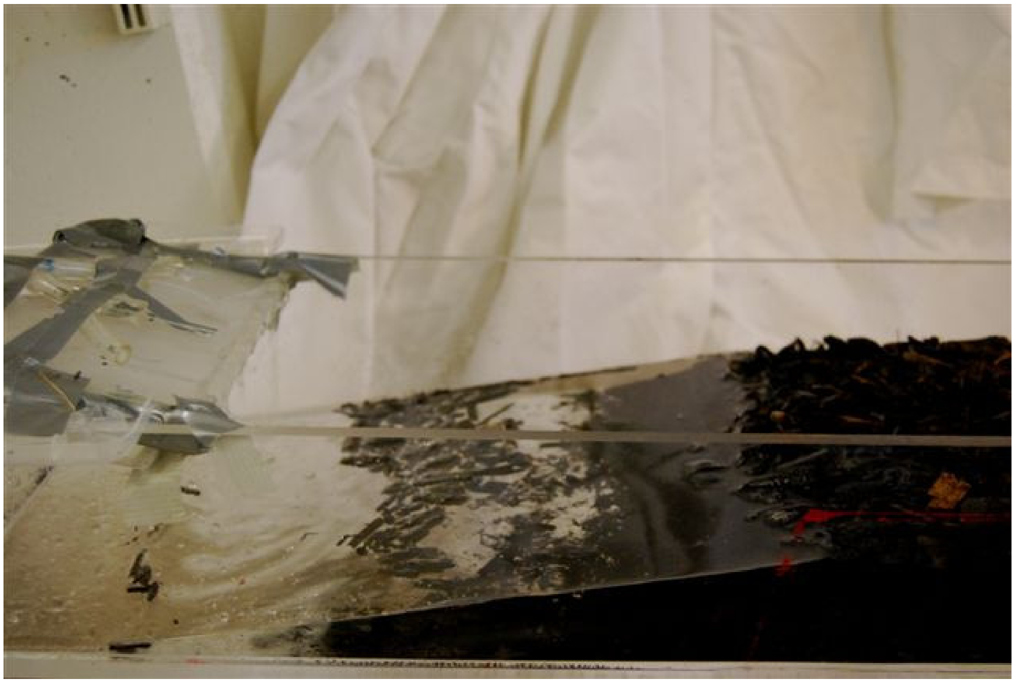


Figure 52- 5 Year Storm Event- 900 ml/min, Compost Only Bed