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Aerobic Attached Growth Biofilter Using Tire Chips And Mixed Broken Glass As Media For Landfill Leachate Treatment

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AEROBIC ATTACHED GROWTH BIOFILTER USING TIRE CHIPS AND MIXED BROKEN GLASS AS MEDIA FOR LANDFILL LEACHATE TREATMENT

by

Daniel Smith, P.Eng, B.Eng (Dalhousie, Hfx, N.S., 2002)

A thesis presented to Ryerson University
in partial fulfillment of the
requirements for the degree of
Master of Applied Science
in the Program of
Environmental Applied Science and Management

Toronto, Ontario, Canada, 2009

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Abstract

Aerobic Attached Growth Biofilter Using Tire Chips and Mixed Broken Glass as Media for Landfill Leachate Treatment

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M.A.Sc.

Environmental Applied Science and Management

Ryerson University

2009

Ontario regulations can necessitate expensive leachate treatment plants in large landfills. Lower-cost technologies may suit rural landfills due to lower waste toxicity and less proximity to residents. One low-cost technology is a trickling filter using tire chips or mixed broken glass (MBG) as filter media instead of non-renewable aggregate.

Aerobic fixed-bed reactors using river-rock gravel, crushed rock, tire chips, and MBG as filter media were compared. COD treatment of stabilized leachate in Phase 1 was limited (max. 21%). Activated sludge was added in Phase 2, and MBG removed 30% COD, 88% BOD₅, 38% NH₃-N, 19% TS, and 100% TSS. All media performed similarly in Phase 3, removing 55-56% COD, 98-99% BOD₅, and 90-98% TSS from a stronger synthetic leachate. NH₃-N removal improved, while TS removal remained limited.

All media types performed similarly, suggesting that tire chips and MBG could be used to treat low-to-medium-strength landfill leachate in a trickling filter.

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Dedication

To Jen for keeping me on my feet.
To my parents for keeping those feet moving.
To Mike and Lisa who are getting married!

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List of Acronyms

AS	Activated sludge	MCTR	Moose Creek Tire Recycling
ASTM	American Society for Testing and Materials	MOE	Ontario Ministry of the Environment
BD	Bulk density	MWW	Municipal wastewater
BOD ₅	5-day biochemical oxygen demand	ND	No data available
C of A	Certificate of Authority	NH ₃ -N	Nitrogen as ammonia
CIWMB	California Integrated Waste Management Board	OD	Outer diameter
COD	Chemical oxygen demand	OECD	Organisation for Economic Co-operation and Development
CP	Chlorophenol	OTS	Ontario Tire Stewardship
CRR	Crushed rock	OWMA	Ontario Waste Management Association
CWC	Clean Washington Center	PCP	Pentachlorophenol
DCP	Dichlorophenol	PD	Particle density
DO	Dissolved Oxygen	PE	Polyethylene
EA	Environmental Assessment	POP	Persistent organic pollutants
EAA	<i>Environmental Assessment Act</i>	PVC	Polyvinyl chloride
EBR	<i>Environmental Bill of Rights</i>	RR	Recirculation ratio
ECO	Environmental Commissioner of Ontario	RRG	River-rock gravel
FRadj	Flow rate after adjustment	RRI	Recycling Research Institute
FRmeas	Flow rate at start of day	SG	Specific gravity
FS	Fixed solids	SMP	Soluble microbial product
HAR	Hydraulic application rate	TCH	Tire chip
HGL	Hamilton Glanbrook Landfill	TF	Trickling filter
HLR	Hydraulic loading rate	ToR	Terms of Reference
HPC	Heterotrophic plate count	TDS	Total dissolved solids
HRT	Hydraulic retention time	TS	Total solids
IC & I	Industrial, commercial and institutional	TSS	Total suspended solids
ID	Inner diameter	UA ³ F	Upflow anaerobic anoxic aerobic floating
IPPC	Integrated Pollution Prevention and Control	VFA	Volatile fatty acid
KVL	Keele Valley Landfill	VOC	Volatile organic compound
MBG	Mixed broken glass	VS	Volatile solids
		WWTP	Wastewater treatment plant

CHAPTER 1: Introduction

A landfill capacity crisis has developed in Ontario over the last ten to fifteen years. The overall number of landfills has decreased from 730 in 1989 to 81 at present (OWMA 2006). This figure includes the landfills serving all three major non-hazardous waste sectors: Municipal; Industrial, Commercial and Institutional (IC & I); and Construction and Demolition. The decline in the number of sites has been attributed to the inability of many operators to avail themselves of the expertise and funds required to meet tightened regulations governing the expansion of existing landfills and the construction of new ones (OWMA 2006). At the same time, it is unlikely that the need for landfill capacity will decline in the province in the near future: Ontario's rate of waste generation increases year by year (Statistics Canada 2005), the province has thus far failed to meet its own waste diversion targets¹ (Miller 2006), there is only one energy-from-waste facility in the province (MOE 2008), and it is becoming increasingly difficult and/or costly to ship waste outside of the province as receiving jurisdictions such as Michigan seek to limit their waste imports (OWMA 2006).

1.1 Leachate Treatment as a Cost-Reduction Target

One significant cost that can be attributed to stiffer landfill regulations is that resulting from the requirement that new and expanded landfills have engineered leachate management systems. Leachate is the percolate from rain and groundwater that passes through and collects at the bottom of a typical solid waste pile (McBean *et al.* 1995). It contains high concentrations of organics, ammonia, metals and other toxic compounds (McBean *et al.* 1995). A leachate leak can contaminate ground-sourced drinking water, poison aquatic organisms, and lead to eutrophication of surface waters (McBean *et al.* 1995). There are thousands of active and inactive waste disposal sites in Ontario (Wellington 2007), and the leachate produced at these sites varies greatly in composition, depending on the age and composition of the site's waste, the season, and the amount of rainfall (Farquhar 1989). As a result, large landfills serving major population centres may require very complex and expensive treatment plants. For example, Green Lane Landfill, now serving the city of Toronto, has an on-site treatment plant that uses physico-chemical, aerobic biological, and anaerobic biological processes to treat its leachate

¹ The latest such target was 60% diversion by the end of 2008 (Miller 2006).

(Toronto 2009). A landfill serving Halifax, Nova Scotia required a leachate treatment plant that cost \$4.2 million to build in 1988 (HRM 2005). Other landfills, such as the Glanbrook Landfill now serving the city of Hamilton, can avoid this cost by relying on local wastewater treatment plants to treat the leachate they produce (Waste Management 2009). However, many small communities do not have centralized wastewater treatment; therefore, new landfills built to serve them would require in-situ treatment. Such landfills built or expanded beyond 40,000 m³ (e.g. – only 0.8 ha given a 5-m bench height) would require a leachate management plan. They often accept only domestic waste, and may be located sufficiently distant from vectors (e.g. – drinking water aquifers) and receptors (e.g. – humans) that they do not pose a significant danger to their communities. For example, the province of Ontario's 1991 landfill inventory (the most recent provincial inventory available, unfortunately) classified many of the then-active landfill sites in Northwestern Ontario as offering "...a low potential for impact on humans..." although they may still have posed a threat to the environment (Ontario 1991, p.192). In these cases, where natural attenuation is not feasible, an affordable leachate treatment technology might be used.

1.2 Trickling Filter as a Cost Reduction Approach

An aerobic trickling filter could be one lower-cost approach to leachate treatment. This technology, used for many years in sanitary wastewater treatment, consists of a packed bed of aggregate media over which is sprayed the wastewater requiring treatment. Biofilms develop on the media and remove contaminants in the fluid through various physical and biochemical processes, such as adsorption and metabolism. Biological treatment can be less costly than equivalent physico-chemical treatment (McBean *et al.* 1995). Aerobic biological systems degrade materials more quickly than do anaerobic systems (Bilgili *et al.* 2008). Mechanical aeration, a significant expense for aerobic systems, can be avoided by relying on the natural air-mixing action of trickling filters.

1.3 Reclaimed Media as a Cost Reduction Approach

Crushed rock is a traditional material for various attached growth filters, including septic fields, landfill drainage layers, and trickling filters. The overall consumption of sand, gravel, and rock aggregates in Ontario amounts to approximately 14 tonnes per person per year (Miller 2006). These aggregates are heavy but low-cost materials, so the cost of transporting them to market is

significant (Miller 2006). Therefore, quarries tend to be sited as close to the markets as possible (Miller 2006). For example, most of the aggregate used in Southern Ontario is taken from the Niagara Escarpment, the Oak Ridges Moraine, and the Carden Plain (Miller 2006). The pressure in these areas to both urbanize available land and preserve sensitive ecological habitats makes it increasingly difficult to start new quarries or expand existing ones (Miller 2006). These pressures could lead to increased aggregate prices, either due to scarcity or increased transportation costs (as aggregate is sourced from farther away).

Plastic packing media is commonly used in modern trickling filters, generally offering better performance than rock media under high organic loading rates (Metcalf and Eddy 2003). Plastic media are also non-renewable and are generally more expensive than crushed rock, although they are lighter and therefore can offer lower construction costs (Martin and Martin 1991). In either case, packing media costs (and aggregate costs in general) could be reduced by substituting reclaimed waste materials for non-renewable aggregates. Re-use of these materials would also support general waste diversion targets.

Two such waste materials that present a particular diversion challenge in Ontario are scrap tires and mixed broken glass (MBG). Scrap tires stockpiled in a landfill or monofill (tires-only stockpile) are a fire hazard and can be a breeding ground for disease-carrying mosquitoes and other pests (Amari *et al.* 1999). The oil and other hazardous compounds released during a tire fire can leach into groundwater (Blackman and Palma 2002). Given the volume of scrap tires generated each year and their associated environmental problems, many countries have taken steps in recent years to reduce existing stockpiles and prevent further accumulation. For example, by 1990 all but two U.S. states had regulations or scrap tire management programs restricting disposal, licensing haulers and processors, and applying advance disposal fees (Merritt *et al.* 2005). By 2001, the American stockpile of approximately 3 billion tires was reduced to 300-800 million tires² (Merritt *et al.* 2005). Ontario has an estimated 5-6 million tires stockpiled, and only 12% of the 10.8 million scrap tires produced each year are diverted for reuse (OTS 2004). A provincial scrap tire management program will be implemented on September 1st, 2009 (OTS 2009). Should it be implemented, one of the keys to its success will be the

² This number was also reduced in part due to some large tire fires that occurred in the 1990's (Merritt *et al.* 2005).

development of sustainable recycling markets for this material. While reuse and retreading are the most energy efficient methods of scrap tire diversion, their markets are hampered by several factors: a relatively small proportion of the scrap tires generated annually are physically suitable for reuse, the public perceives retreaded tires to be less safe than virgin tires, and producers of retreaded tires have difficulty competing with the price of low-cost tires imported from Asia (Amari *et al.* 1999; OECD 2006). Instead, as Table 1-1 indicates, in North America a significant portion of tires are burned for energy ("Energy Recovery"), a practice that raises environmental concerns and recovers only 40% of the energy put into making the tires (Amari *et al.* 1999).

Table 1-1. Prevalence of options for used tire processing (Adapted from OECD 2006).

	European Union Average (2003)	United States Average (2000)
Scrap Tires Generated (1,000's of Tons)	2,739	3,600
Reuse (% of Recovered)	2.3	11
Export (% of Recovered)	7.7	5
Retreading (% of Recovered)	8.9	9
Material Recycling (% of Recovered)	26.6	29
Energy Recovery (% of Recovered)	22.0	34
Landfill & Other Disposal (% of Recovered)	32.5	12
Overall Used Tire Recovery (%)	67.5	N/A

Material recycling is the most common alternative to energy recovery; however, recycling markets for scrap tires are difficult to sustain due to high processing costs and low commodity value (OECD 2006). Recycling market diversification through the development of new applications, such as leachate drainage layer aggregate or a filter media for leachate treatment, would foster increased diversion.

McKenzie (2003) noted that using tire chips instead of gravel can reduce aggregate cost by anywhere from 10% to 90%. A basic cost comparison between tire chips and crushed rock is illustrative. Tire chips are available in Ontario for approximately \$30/tonne plus the cost of transport (Villeneuve 2008), and they have approximately 1/3rd the bulk density (BD) of crushed rock (Grimes *et al.* 2003). Therefore, tire chips can be purchased for the equivalent of \$10/tonne

of crushed rock³. Crushed rock price quotations obtained for comparison from the Greater Toronto Area ranged from \$12/tonne - \$20/tonne freight-on-board (i.e. – before shipping cost) (Cummings 2008). Over similar transport distances, savings in shipping costs would also be realized given the lower per-truck weight of tire chips.

The Waste Diversion Act of Ontario requires municipalities to collect recycled glass in the Blue Box program. Aside from beer, wine, and spirit containers, which are recovered through dedicated diversion programs, 53,743 tonnes of glass were recovered by Ontario municipalities in 2006 (Stewardship Ontario 2007). Much of this is color-mixed and broken, making it more difficult and costly for glass recyclers to use. Therefore, municipalities incur a cost to market this mixed broken glass (MBG) (Stewardship Ontario 2007). For example, Toronto produces approximately 20,000 tonnes per year of glass and pays an average of \$48.75/tonne to have it recycled, for a cost of almost \$1 million per year (Spears 2008). A new MBG processing facility built by Unical to serve the GTA may lower this cost to \$10/tonne (Spears 2008); however, the use of glass in a municipality's own civil works could reduce the cost even further.

From the aggregate substitution perspective, MBG is already available at municipal transfer stations and/or sorting centres; separate additional hauling contracts to bring aggregate in from outside could be eliminated. If grading were required, this expense could be at least partially offset by not having to pay tipping fees to recyclers.

1.4 Hypothesis

Both scrap tires and MBG have the potential to be suitable aggregates for biofilm growth. Scrap tires, chipped into aggregate form, are relatively inert due to the vulcanized polymers they are comprised of, and have rough surfaces conducive to microbial attachment. MBG is also suitably inert, and while it has smoother surfaces than tire chips, it offers the benefit of being easy to process into well-graded material. Therefore, this thesis began with the following hypothesis:

³ This assumes negligible compression of the tire chips, an assumption that can be made for a trickling filter, but not for a leachate drainage layer laid under several meters of solid waste. Therefore, the "true" bulk density (and thus the cost per unit area) of tire chips in a drainage layer would be somewhat higher.

Tires and mixed broken glass are suitable substitutes for gravel as a packing media in a trickling filter used to treat landfill leachate.

1.5 Objective

While there have been many studies on the suitability of tire chips for landfill drainage layers and septic fields (e.g. – Grimes *et al.* 2003, Phaneuf and Glander 2003, Warith and Rao 2004, Rowe and McIsaac 2005), there is little on their use in aerobic biofilters, particularly in comparison to other media. Similarly, while MBG has been studied for use in landfill blankets and sanitary wastewater treatment, there is little research on its use to treat landfill leachate, and none on using it in a “raw”, non-graded form. Using non-graded MBG would save the cost of processing. This may or may not be a trivial cost, depending on how much glass would be required for the application. For example, while the cost of a glass crusher may only be a few thousand dollars (e.g. - The Bore-lee Glass Equipment 2009), larger applications may require the use of heavy mobile equipment and associated labour (e.g. – Anon. 1994). Therefore, the objective of this project was:

To compare the performance of tire chips and mixed broken glass to that of a common, non-renewable aggregate (crushed rock) as a packing media in an aerobic biofilter used to treat landfill leachate.

For this study, leachate of low to moderate strength was used, as might be found in old rural landfills requiring the installation of a leachate management system in order to be expanded.

1.6 Scope of Work

The scope of this work included:

- design, construction, and operation of three laboratory-scale aerobic fixed-film bioreactors using the three test media;
- sampling and measurement of the leachate circulated through these reactors for pertinent variables; and
- examination of each reactor at the end of the test phases for evidence of biofilm accumulation and/or clogging.

CHAPTER 2: Literature Review

The literature was reviewed from both an environmental management (the “why”) and an applied science (the “how”) perspective in order to best define the appropriate methodology for this experiment. Ontario’s landfill regulations were reviewed, particularly as they apply to leachate. The benefits and disadvantages of aerobic fixed-film filters were studied, as were the conventional and experimental packing media in use or under study. The difficulties of recycling tires and MBG were researched, as were the problems associated with failure to recycle them and the benefits they might offer as biofilter packing media. Literature related to the use of each of these media in similar applications was reviewed.

2.1 Landfill Regulations

Ontario’s *Environmental Assessment Act (EAA)* requires any development activity classified as a new major commercial or business enterprise or activity to submit both the terms of reference (ToR) for an environmental assessment (EA), and the EA itself to the Ministry of Environment (MOE) for approval. Public consultations must be held on the ToR and the EA. In addition, the *Environmental Bill of Rights (EBR)* statute grants citizens the right, among others, to appeal certain Ministry decisions, including those regarding EAs (Miller 2006). The Ontario Waste Management Association (OWMA 2006) has claimed that such EA challenges regarding new landfill proposals have resulted in few new landfill approvals in the last ten to fifteen years. The provincial government has attempted to address this problem by creating the *Waste Management Projects Regulation* (O. Reg 101/07) under the *EAA*, which provides a streamlined assessment and public consultation process for medium-sized (40,000 – 100,000 m³) landfill projects⁴.

While O. Reg 101/07 may enable landfill proponents to reduce the cost of public consultation, there remain the costs associated with designing and maintaining the site to meet more stringent environmental standards. Landfill design and operation is regulated under Ontario’s *Environmental Protection Act* by the issuance to site owners of Certificates of Authority (Cs of A), which are subject to one of two regulations. Prior to 1998, landfills operating in the province

⁴ The Environment Commissioner of Ontario (ECO) has criticized this regulation as limiting the extent of public involvement in such projects, and therefore defeating the original intent of the *EAA* (Miller 2008). In addition, the regulation tends to favour incineration projects, and the ECO questions the justification of this approach (Miller 2008).

were subject to the *General - Waste Management Regulation* (R.R.O. 1990, Reg. 347). O. Reg. 347/90 does not prescribe specific effluent variable limits. Instead, it uses qualitative language that permits considerable interpretive flexibility. For example, landfill surface drainage cannot “adversely affect” neighbouring properties or natural drainage. To provide more stringent requirements, the *Landfilling Sites Regulation* (O. Reg 232/98) was introduced to supercede O. Reg. 347/90 for all new sites or site expansions that would result in a fill capacity exceeding 40,000 m³. O. Reg 232/98 requires that a report be submitted to the MOE detailing the expected leachate quality and quantity, the liner system, the leachate management system, and the contingency plans to be followed should these systems fail. Means for determining the minimum quality of discharges to surrounding groundwater and surface water are specified, as are monitoring variables and frequency.

Existing rural landfills are typically subject to the less stringent O. Reg. 347/90, and rely on natural attenuation to treat leachate. The communities that operate these sites lack the funding and technical expertise to design and construct the leachate management systems needed to meet O. Reg 232/98. Therefore, when their existing landfills close, these communities typically ship their waste to sites operated by larger population centres⁵. For example, consider Northwestern Ontario: 19 landfills in this region will be full by 2015 (Davies and Kamp 2005). One of these, Trout Lake Landfill serving Gorham and Ware townships, closed in 2007-2008 (Davies and Kamp 2005). Unable to afford a new or expanded site, the townships contracted to ship their waste to the John Street Landfill in Thunder Bay (Davies and Kamp 2005)⁶. Thunder Bay was able to complete the EA process to expand John Street (Thunder Bay 2008).

The John Street Landfill arrangement highlights the fact that waste disposal self-sufficiency (or “waste independence”) is becoming an exclusive asset. Finding ways to reduce the capital cost of meeting O. Reg 232/98 could enable more communities to acquire that asset.

⁵ Closing old landfills and shipping waste out of town is not the end of the story for small communities (or large ones) because the *Environmental Bill of Rights* permits possible legal action by citizens regarding contamination from these “legacy” sites (Dahme and Lococo 2003).

⁶ Another option for small municipalities is to attempt to expand their current landfill capacity by asking for changes to their site C of As that will allow them to remain under O. Reg. 347/90. For example, the MOE approved an amendment to the C of A for the Edwards Landfill in Haldimand-Norfolk County, allowing it to increase its fill rate from 10 tonnes/day to 500 tonnes/day (Miller 2006). Not surprisingly, this raised serious public environmental concerns, and local residents successfully appealed the decision under the *EBR*; a hearing is now underway (Miller 2006).

2.2 Characteristics of Leachate

Table 2-1 compares some of the variable values of the now relatively stabilized Keele Valley Landfill (KVL) leachate to those of KVL leachate peak concentrations and of a typical untreated municipal wastewater. The values of conservative variables such as chloride concentration increase over the life of a landfill to many times that of a typical municipal wastewater (MWW). The values of non-conservative variables such as Chemical Oxygen Demand (COD) and Ammonia concentration ($\text{NH}_3\text{-N}$) reach a peak before declining to levels that are still much higher than those of MWW. The upshot of these comparisons is that methods used to treat typical wastewater must first be evaluated for their ability to handle significantly higher pollutant loads before they can be applied to leachate treatment.

Table 2-1. Comparison of selected variables: leachate peak, leachate stabilized, & municipal wastewater.

Variable (mg/L except for pH)	KVL Peak (Rowe 1995)	KVL Stabilized (Benda 2008)	Strong Municipal Wastewater (Reynolds and Richards 1996)
COD	17,116	1,975	910
BOD	12,367	ND ⁷	400
$\text{NH}_3\text{-N}$	ND ⁷	1,260	35
TSS	ND ⁷	33	360
Chloride	2,247	4,000	83
pH	6.3	7.90	ND ⁷

Leachate treatment is complicated not only by the strength of leachate constituents, but by their variety and variability over time. These characteristics are governed by variables such as the age of the waste (or more specifically, what phase of decomposition it is in), its composition, the amount of moisture present, the depth of the waste pile, and its temperature.

2.2.1 Phases of Decomposition

Leachate develops when percolating water mobilizes contaminants from refuse through dissolution or suspension (Farquhar 1989). Increased moisture content promotes microbial activity, which contributes metabolic byproducts such as volatile fatty acids (VFAs) to the leachate (Farquhar 1989). This leachate creation continues long after a landfill has closed

⁷ ND = No data available

(Farquhar 1989). According to McBean *et al.* (1995), solid waste decomposition occurs in three phases. Phase 1 is aerobic decomposition, a short-lived phase (typically less than one month duration⁸) that ends once available oxygen is depleted (McBean *et al.* 1995). Up to 20% by volume of the gas produced in this phase is hydrogen, particularly in dry sites (McBean *et al.* 1995).

Phase 2 is acetogenic anaerobic decomposition, where facultative and anaerobic bacteria hydrolyze and ferment cellulose and other organics into VFAs and other simple, soluble compounds such as alcohols (Farquhar 1989; McBean *et al.* 1995). Lasting years or decades, this phase produces leachate with high Biochemical Oxygen Demand (BOD) values (often upwards of 10,000 mg/L), high BOD/COD ratios (often greater than 0.7), high ammonia concentrations (500-1,000 mg/L), low pH (5-6) resulting from VFA production, high levels of dissolved metals due to the low pH, and strong odors (McBean *et al.* 1995). Carbon dioxide is the main gas produced; methane and hydrogen are also generated (McBean *et al.* 1995). BOD indirectly represents the degradable organic carbon content in the leachate. If discharged untreated into the environment, this overabundance of organic carbon can over-stimulate microbial growth, leading for example to eutrophication of surface waters. High ammonia concentrations result from hydrolysis and fermentation of biodegradable nitrogenous compounds (Onay and Pohland 1998). Ammonia is an environmental pollutant that is toxic to aquatic life when present in a water body in concentrations upwards of 0.3 – 3.0 mg/L, depending on environmental factors such as pH, salinity, and temperature (Ruffier *et al.* 1981).

In Phase 3, methanogenic bacteria consume the simple organic compounds produced in Phase 2, producing carbon dioxide and methane (McBean *et al.* 1995). Phases 2 and 3 overlap and arrive at a dynamic equilibrium over many years, as methanogenic bacterial communities slowly develop (McBean *et al.* 1995). Leachate from Phase 3 is termed “stabilized”, although biological activity in the landfill is at its peak (McBean *et al.* 1995). Relatively high levels of landfill gas are typically produced for several years, after which low-level production can continue for many decades, until most of the waste is decomposed and air can once again penetrate the pile (McBean *et al.* 1995). Phase 3 leachate has low BOD values (in the hundreds

⁸ Phase 1 can last longer in shallow waste cells, where air can continue to penetrate the pile (McBean *et al.* 1995).

of mg/L) and low BOD/COD ratios, but continued acetogenic activity maintains ammonia concentrations, and metals and anions (iron, chlorides, sulfates, etc.) can continue to leach slowly over many years (McBean *et al.* 1995). Non-degradable COD remaining in such leachate consists of organic substances of high molecular weight that are difficult to biodegrade (humic and fulvic compounds), soluble microbial product (SMP) from biological activity, and nondegradable inorganic matter (Bilgili *et al.* 2008; Diamadopoulos 1994).

2.2.2 Refuse Composition

Refuse composition defines the sort of organic materials available for biodegradation, as well as the toxic substances (such as pesticides and herbicides) and metals present (McBean *et al.* 1995). As new cells are added to the landfill over time, the leachate produced becomes a composite of readily biodegradable compounds from new refuse mixed with slowly degradable compounds from old refuse, in addition to minute but potentially hazardous concentrations of metals and priority pollutants (Farquhar 1989).

2.2.3 Moisture Content

Moisture addition affects the influence of solute dissolution, biological activity, and particle entrainment on leachate quality (McBean *et al.* 1995). For example, anaerobic microbial activity tends to have a larger influence on organic strength at low flow rates, while flushing by high flows reduces this influence (McBean *et al.* 1995). Flow rates are dependent on climate, season, landfill size, and hydraulic conductivity. For example, in temperate climates flows tend to be reduced in the summer months due to changes in precipitation, temperature, and solar radiation. Increasing the landfill size will increase the flow, while the addition of less permeable final cover (reducing hydraulic conductivity) will increase runoff. These changes in flow can have a significant impact on treatment systems.

2.2.4 Refuse Temperature and Depth

Refuse temperature influences leachate character by affecting the rate of biological and chemical activity in the waste. Salt solubility and the kinetic rates of many chemical reactions increase with temperature. For example, Knox (1985) recommended storing winter leachate for treatment in summer months, when treatment kinetics would be better. Alkalinity and dissolved oxygen

concentration are also impacted (Clesceri *et al.* 1998). Meanwhile, increased refuse depth allows percolating leachate more time to approach its solubility limit, increasing leachate strength and requiring more time to achieve stability. From a biological standpoint, different species of microorganism will grow within different temperature ranges, depending on whether they are psychrophilic (0 – 10°C), mesophilic (10 – 45°C), or thermophilic (45 – 75°C) (Reynolds and Richards 1996). Landfill temperatures are typically suitable for mesophiles, and can increase to the thermophilic range (McBean *et al.* 1995). Physical characteristics such as turbidity and colour are affected by temperature as a result of its impact on chemical and biochemical processes.

2.3 Biological Leachate Treatment

If landfill leachate is discharged to a sanitary sewer, some degree of pre-treatment may be necessary, depending on the requirements of the receiving sanitary wastewater treatment plant. In Ontario, if leachate is discharged directly to the environment, on-site treatment is necessary unless a site-specific natural attenuation design can be shown to meet O. Reg 232/98.

Physico-chemical treatment processes such as chemical precipitation, chemical coagulation-flocculation, powdered activated carbon adsorption, chemical oxidation, membrane filtration, air stripping for ammonia, or ozonation can effectively treat a strong, variable effluent such as MSW leachate (Diamadopoulos 1994). However these methods typically require costly and potentially hazardous chemicals and often produce large amounts of toxic sludge that require disposal (IPPC 2007). Biological treatment minimizes these costs by relying on microorganisms to degrade and/or adsorb contaminants (McBean *et al.* 1995). Sludges are also produced in biological processes, but these can be recycled and/or retained to improve degradation, and chemical requirements are limited to nutrients and pH buffers (McBean *et al.* 1995). Shipin *et al.* (2007) achieved performance similar to that of an equivalent physico-chemical system at 1/20th of the cost, using an upflow anaerobic/anoxic/aerobic floating (UA³F) filter.

There are limitations to biological treatment. Diamadopoulos (1994) noted that leachate with a low BOD:COD ratio (e.g. – less than 0.1) is not readily treated by biological means, as it is already stabilized. Further removal of non-biodegradable COD usually requires the use of

physico-chemical methods. In addition, it is common for leachate to be very saline, which may require several months of acclimatization by bacteria used for treatment (IPPC 2007). In addition, treatment of acetogenic leachates containing BOD and COD concentrations above 5,000 mg/L may lead to overgrowth of biomass that results in clogging or sludge build-up problems. Similarly, precipitation of high concentrations of iron, calcium, and other metals may cause build up of inorganic sludge and deposits (IPPC 2007).

2.3.1 Aerobic Biological Treatment

The majority of successful biological treatment systems incorporate an aerobic process (IPPC 2007). Aerobic degradation has been found to effectively reduce the high COD, BOD, and $\text{NH}_3\text{-N}$ concentrations in acetogenic leachates, as well as reducing the $\text{NH}_3\text{-N}$ concentrations in methanogenic leachates (IPPC 2007). While a high COD concentration may sometimes remain in treated effluent, it is primarily inert and found to have little toxicity (IPPC 2007). For example, Bilgili *et al.* (2008) found that the ratio of non-biodegradable COD to total COD was higher in leachate from aerated waste columns than in leachate from anaerobic columns (60% vs. 50%), and concluded that this was the result of more (non-degradable) soluble microbial product (SMP) generated by aerobic biological activity. Bilgili *et al.* (2008) also found that aerobic degradation of leachate requires a shorter hydraulic retention time (HRT) than does anaerobic digestion to achieve equivalent treatment. Other advantages to aerobic treatment compared to anaerobic treatment include lower temperature requirements, better ammonia removal, and less sensitivity to acidic conditions (McBean *et al.* 1995). Aerobic bacteria can also treat many trace organic compounds, often with complete or near-complete degradation (IPPC 2007). These contaminants are often actually degraded rather than concentrated in a sludge or brine (IPPC 2007). Similarly, high transfer rates of some metals and metal-like compounds into the biomass can be achieved (e.g. – 70% removal of arsenic) (IPPC 2007).

The amount of sludge that is produced under aerobic conditions will vary depending on leachate age. As bacteria grow, some of the organic compounds present in the leachate are used in cell synthesis. Methanogenic leachate has a relatively low organic content, so cell lysis and death tend to balance cell synthesis, resulting in little net increase in bacterial population (and hence little sludge development) (IPPC 2007). Acetogenic leachate will promote the growth of more

biomass; however this sludge provides the benefit of consuming a small portion of ammoniacal nitrogen. For example, Robinson and Maris (1983) found nitrogen uptake in cell growth accounted for approximately 3.6% of COD reduction.

Three processes are typically included in aerobic treatment systems: biological oxidation of organic compounds to carbon dioxide and water, nitrification, and (anaerobic) denitrification (IPPC 2007). Under aerobic conditions, autotrophic bacteria (those that use inorganic carbon as a food source) derive energy from the oxidation of ammoniacal nitrogen. This nitrification process takes place in two stages via two distinct bacterial groups. In the first stage, ammoniacal nitrogen is oxidized to nitrite by the genus *Nitrosomonas* according to the following equation (IPPC 2007):

Nitrite is then oxidized to nitrate by a species of *Nitrobacter* as follows (IPPC 2007):

Establishing stable nitrification can be challenging. For every 1 kg of ammoniacal nitrogen that is nitrified, 4.27 kg of O_2 is consumed, 7.14 kg of alkalinity (as CaCO_3) is destroyed, and 0.22 kg of new cells (sludge) is synthesized (IPPC 2007). Nitrifying bacteria are more sensitive to environmental conditions than are carbon-oxidizing bacteria, and the most typical causes of nitrification failure are insufficient alkalinity or dissolved oxygen (DO). Nitrification lowers pH, and nitrifiers can be easily inhibited by a pH below 6.5 (a pH between 7.5 and 8.0 is optimal). A strong methanogenic leachate may require the addition of alkalinity of up to 50% of the value present in the raw leachate (IPPC 2007). A DO concentration below 2.0 mg/L is also inhibitory (IPPC 2007). Other inhibitors include toxic constituents and temperatures below 5°C or above 35°C (IPPC 2007). The latter inhibitor is important to note in temperate climates, where cold temperatures from diurnal and seasonal effects can inhibit nitrification (IPPC 2007).

Phosphorous and micronutrients may also be required (IPPC 2007).

Too much ammoniacal nitrogen or nitrite is toxic to nitrifiers and can lead to a nitrite build-up (IPPC 2007). Levels of $\text{NH}_3\text{-N}$ above 80 mg/L can create toxic inhibition; landfill leachates typically have concentrations of at least 500 mg/L, and often above 1,000 mg/L (IPPC 2007). Therefore, a means of reducing the shock of these concentrations must be considered. This toxicity is related to free ammonia, which is a function of ammoniacal-N concentration and pH (the higher the pH, the more ammoniacal-N exists as free ammonia) (IPPC 2007).

Nitrification transforms nitrogen from ammonia to nitrate forms at close to a 1:1 ratio, but it does not actually remove it from the effluent. Nitrate is the limiting nutrient for photosynthetic autotrophs. High concentrations of nitrate in receiving waters can stimulate algal growth, which will increase the growth of microorganisms that consume algae, possibly leading to eutrophication of the water body. Therefore, both nitrification and denitrification steps are often included in leachate treatment processes to remove nitrogen from effluent water before it reaches a receiving water body. Denitrification, the most widely used nitrogen removal process in municipal wastewater treatment, is the reduction by facultative heterotrophic organisms of nitrate to nitrogen gas under anoxic conditions (no molecular oxygen). These organisms use nitrate as the electron acceptor and organic carbon as an energy and food source, according to the following equation (IPPC 2007):

The denitrification process requires a consistent organic carbon source (i.e. - a consistent oxygen demand), which usually entails the addition of methanol (IPPC 2007). For each kg of nitrate that is denitrified, 2.47 kg of methanol are required, 0.45 kg of new cells are synthesized, and 3.57 kg of alkalinity are formed (IPPC 2007). Denitrification is inhibited by DO concentrations greater than 0.2 to 0.5 mg/L, by low temperatures, and by temperatures above 40°C (IPPC 2007). Denitrification also requires a narrow pH range (7 – 7.5), and produces alkalinity, so the addition of acid may be required (IPPC 2007).

2.3.2 Attached Growth Aerobic Reactors

Aside from the presence or lack of oxygen, biological treatment systems are also categorized by whether microorganisms are suspended (free-swimming) in the wastewater or instead grow as a

biofilm attached to a packing media and used to filter the wastewater. The media does not contribute directly to treatment; rather it serves simply as a surface for biofilm attachment (IPPC 2007). Successful application of suspended systems requires the development and acclimatization of biomass that flocculates, settles, and compacts well (IPPC 2007). In suspended sludge systems there is a risk of sludge washout due to poor settling conditions (McBean *et al.* 1995). Washout degrades effluent quality and represents a loss of bacteria, whereas microorganisms (nitrifying bacteria in particular) can affix themselves strongly to a media in attached-growth systems. Biomass does slough off of filter media after the organisms closest to the media surface die from lack of oxygen due to film thickening. However this sloughed-off material is easier to manage than poorly conditioned sludge.

There is conflicting literature about whether attached-growth technologies are suitable for leachate treatment. IPPC (2007) claimed that in the UK there are few examples of successful leachate treatment using trickling filters and many examples of failure. They asserted that the small pore volumes and short hydraulic retention times associated with biofilters do not permit complete mixing, and so chemical shock loadings, such as elevated $\text{NH}_3\text{-N}$ concentrations, cannot be rapidly diluted (IPPC 2007). For example, nitrifier inhibition at the inlet of a downflow aerobic reactor due to high $\text{NH}_3\text{-N}$ concentrations (>80 mg/L) can spread down through the media column (IPPC 2007). However McBean *et al.* (1995) stated that attached growth reactors offer better tolerance of hydraulic and chemical shock loading. Similarly, Knox (1985) noted that an activated sludge (AS) pilot plant under test at a UK landfill suffered from flotation problems due to high $\text{NH}_3\text{-N}$ concentrations in the leachate; a fixed-film trickling filter plant operating in parallel to the AS plant performed well by comparison.

A similar apparent conflict exists regarding the effect of high oil and grease content in the wastewater. IPPC (2007) stated that fixed microorganisms can become "blinded" by the presence of oil and grease, whereas these constituents tend to be emulsified and degraded in suspended systems. However Del Pozo and Diez (2003) observed that the high oil and grease content in slaughterhouse wastewater caused flotation problems (poor settling of solids) in suspended-biomass reactors; therefore fixed-film reactors are preferable.

Failure to achieve complete mixing can cause difficulties in evenly distributing oxygen and alkalinity (IPPC 2007). Lack of alkalinity control leads either to local scale formation when alkalinity is too high, or to inhibition when it is too low (IPPC 2007). Similarly, locally acidic pH at areas of high nitrification can become inhibitive, and pH throughout the reactor column cannot be monitored by a single probe the way it can be in a suspended treatment tank (IPPC 2007). Therefore, it can be very difficult to direct appropriate doses of alkalinity to the point of need within fixed-film reactors when treating stronger leachates (IPPC 2007). Conversely, simply adding alkalinity at the inlet to buffer acidity will release greater levels of toxic free ammonia (IPPC 2007).

2.3.3 Trickling Filters

Fixed-film reactors can offer a cost advantage over activated sludge reactors under aerobic conditions: AS reactors require mechanical aeration, which can account for up to half of the electricity usage of a sanitary wastewater treatment plant (Metcalf and Eddy 2002). By comparison, a trickling filter (TF), an aerobic fixed-film reactor over which wastewater percolates by gravity, can use natural aeration instead (IPPC 2007). The media bed for a TF is typically 2 – 4 m deep and can be made of a variety of materials (IPPC 2007). Wastewater is irrigated evenly over the filter surface, and effluent from the bottom is passed through a clarifier to remove biological solids that slough off from the media (IPPC 2007). In terms of footprint, trickling filters are fairly similar to activated sludge systems. The TF system discussed by Knox (1985) included the filter, with an area of 6.2 m^2 and a volume of 16.5 m^3 , and a humus tank with an area of 3.5 m^2 . An activated sludge system installed for comparison included an aeration tank with a volume of 8.07 m^3 , a clarifier with an area of 5.5 m^2 and a volume of 9.3 m^3 , plus a sludge holding tank (Knox 1985).

TFs are subject to many of the general disadvantages of attached-growth filters. They may require regular backwashing to prevent clogging from organic and inorganic sludge (Ball 1998; McBean *et al.* 1995). However Knox (1985) did not see any evidence of clogging, either organic or inorganic, over two years of treating a stabilized leachate in-situ. Clogging in high and medium strength leachates is often due to oxidation and precipitation of iron and manganese (Knox 1985). These metals are not prevalent in stabilized methanogenic leachates, having

already been leached out; however, calcium carbonate can be (Knox 1985). The transport of leachate from within the landfill to the surface results in a change in pressure that allows CO_2 to escape (Knox 1985). This raises the leachate pH, changing the carbonate-bicarbonate equilibrium and resulting in calcium carbonate precipitation (Knox 1985). The use of a collection lagoon or trench with sufficient hydraulic retention time upstream of the treatment system may allow precipitation and pH stabilization of the leachate before it enters the trickling filter (Knox 1985).

TFs may not uniformly distribute concentrations of leachate components such as ammonia (IPPC 2007). However Knox (1985) noted that leachate with a low $\text{BOD}/\text{NH}_3\text{-N}$ ratio would produce poorly settled effluent in an activated sludge reactor due to the presence of free-swimming nitrifiers; therefore, he recommended using trickling filters for such a leachate.

2.4 Trickling Filter Design Considerations

Aeration, temperature, establishment of the biofilm prior to full operation, hydraulic and contaminant loading rate, hydraulic retention time, recirculation ratio, influent distribution, sludge collection and backwashing, and packing media must all be considered as variables in the design of a suitable trickling filter.

2.4.1 Aeration

The method of providing sufficient oxygen for aerobic growth depends largely on the system design. For example, trickling filters typically rely on natural aeration. In a study of wastewater treatment by trickling filter, Yang (2002) provided aeration by natural convection via an opening beneath the reactor column; air from the relatively warm reactor rises, drawing in cooler air. To permit sufficient natural aeration, Ball (1998) recommended ensuring that the filter is not saturated with water. Upflow attached-growth reactors, which improve mixing through the column, require mechanical aeration from the bottom. The position of the diffuser tube in such a reactor is important. Xie *et al.* (2004) noted that placing the diffuser tube 100 mm above the bottom of their lab reactor resulted in more even distribution of the air through the filter media, compared to placing the tube at the very bottom.

Gálvez *et al.* (2003) noted that the nitrogen removal rate improved as the aeration rate in their reactor was increased from 7.8 to 25.6 m³h⁻¹m⁻². While COD removal improved significantly as airflow was increased from 7.8 to 13.76 m³h⁻¹m⁻², little improvement occurred beyond that; therefore, oxygen concentration was no longer the limiting factor for heterotrophic bacterial activity (Gálvez *et al.* 2003). TSS removal changed very little with change in oxygen (Gálvez *et al.* 2003). This is not surprising given that settling and adsorption to the media and biofilm would be the primary means of TSS removal.

2.4.2 Temperature

Much of the lab research on attached growth filters has been conducted at ambient lab temperatures, typically between 20 – 25°C (warmer when research is conducted in warm climates) (e.g. – Clabaugh 2001, Yang 2002, Xie *et al.* 2004). Little in-situ research has been conducted in temperate climates, where winter temperatures can seriously impair treatment. Knox's (1985) two-year study, discussed previously, took place over two years at the Pitsea landfill in the UK. He found that the reactor froze completely for a period of 6 weeks over December and January (Knox 1985). He also found that the influent NH₃-N concentration tended to peak over the winter months (Knox 1985). Comparing the TF to the AS reactor, he found that filter effluent temperature was typically 1 – 2°C cooler than that of the AS reactor, and attributed the difference to the heat input from the blowers in the sludge plant and the natural cooling effect of the TF (Knox 1985). In addition, Knox (1985) noted that the TF followed diurnal temperature variations much more closely than did the AS plant. Knox (1985) presented a table of NH₃-N, BOD, and TSS influent and effluent concentrations vs. temperature. Interpretation of the table indicates that NH₃-N removal did worsen with decreasing temperature, although after dropping off below 16.0°C it remained fairly steady. Interestingly, BOD removal varied considerably and did not seem to relate to temperature change, indicating that other factors were more influential. While TSS does appear to have gradually increased, the data are complicated by the fact that effluent values were occasionally higher than influent values, probably due to biomass sloughing. Similarly, a biological filter study conducted by Mann *et al.* (1998) reported that a temperature shock caused by rapidly chilling influent sanitary wastewater from 8°C down to 4°C appeared to have a greater impact on nitrification than it did on carbonaceous matter removal.

2.4.3 Establishment of Biofilm

Acclimation of the microbial population to the leachate being treated is critical to the development of a successful biofilm. Acclimation rate varies greatly from case to case, depending on the source of the microbes, the substrate they are initially fed to establish the biofilm, the hydraulic loading rate and the temperature. It is ideal to use microbes that are indigenous to the leachate, but this is not always possible. Frequently, activated sludge from a sanitary wastewater treatment plant is used, particularly during laboratory-scale studies. In such a case applying leachate immediately could shock and kill off the bacteria in the sludge. Instead, a diluted mixture of the leachate should be used, often preceded by a nutrient solution to promote growth.

For testing trickling filter treatment of a simulated wastewater containing polypropylene glycol methyl ether diluted with distilled water, Yang (2002) established an initial biofilm by first trickling a nutrient substrate through the reactor. Yang (2002) achieved sufficient growth after 21 days. Conversely, Gálvez *et al.* (2003) achieved steady-state COD and TSS removal within 3 days, and steady-state nitrification within 13 days, using only sanitary wastewater (no nutrient substrate).

2.4.4 Loading Rates, Hydraulic Retention Time, and Recirculation Ratio

The BOD removal performance of a trickling filter has been shown to correlate well with the volumetric BOD loading rate; that is, the mass flow rate of the contaminant per unit of reactor working volume (Metcalf and Eddy 2002). Aside from toxic effect, a finite volume of microorganisms has a limit to how much it can metabolize. Dermou and Vayenas (2007) found that increased influent chromium concentration greatly increased the time for complete reduction. The lowest influent concentration tested, 5 mg/L, resulted in a treatment time of approximately 45 min, while the highest concentration, 100 mg/L (or five times greater), required over 350 min (almost eight times greater) (Dermou and Vayenas 2007).

Hydraulic loading rate (HLR) is the volumetric flow rate through (perpendicular to) the cross-sectional area of a reactor (Reynolds and Richards 1996). Hydraulic retention time (HRT) is the amount of time between when a fluid element enters and leaves a reactor (Reynolds and

Richards 1996). HLR has a varied effect on treatment performance. Generally, an HLR that is too high can cause cell damage and shearing of the biofilm, toxic shock from the associated contaminant loading rates, and an HRT that is too short to permit sufficient contaminant degradation (Reynolds and Richards 1996). Yang (2002) determined an empirical relation between BOD removal and HLR for a wastewater trickling filter; BOD removal decreased as HLR increased. Gálves *et al.* (2003) varied HLRs applied to an aerobic/anoxic combined system from 0.35 to 1.59 $\text{m}^3\text{h}^{-1}\text{m}^{-2}$, and found that nitrogen removal decreased as HLR increased to 0.71 $\text{m}^3\text{h}^{-1}\text{m}^{-2}$ (equivalent to an HRT of 1.6 h in their system), beyond which only negligible nitrogen removal occurred. COD removal was somewhat sensitive to HLR, while TSS was less so (Gálvez *et al.* 2003). Knox (1985) applied HRTs ranging from 15 h (summer) to 4.5 days (winter) to the pilot TF discussed earlier. HRT can be extended by increasing the recirculation ratio (RR). Recirculation has been found to best benefit low-rate systems, where it provides increased flow to improve wetting and flushing of the packing media (Metcalf and Eddy 2002).

One method of controlling HLR is through dosing; that is, the application of pulses of leachate rather than a continuous flow. In a discussion on sand and packed media filters for septic treatment, Ball (1998) recommends applying small, frequent doses at a suitable hydraulic application rate (HAR, the volume of wastewater applied per dose). In the case of leachate treatment, a suitable HAR would depend on the fluid volume requiring treatment and the upstream storage capacity as well as the effect on treatment performance.

2.4.5 Influent Distribution, Sludge Collection, and Backwashing

As discussed previously, difficulty in achieving complete mixing can be a disadvantage of attached-growth filters. Uniform distribution of the influent over the entire filter will help to address this. Typically a rotary distributor arm is used for TFs. For small-scale reactors, a manifold plate with small, evenly spaced holes is suitable (Ball 1998; Reynolds and Richards 1996). Provisions for sludge and clog removal must also be provided. An underdrain, typically of gravel or crushed stone, is useful for collecting sludge, including the biomass that occasionally sloughs off of the filter (Mondal and Warith 2008).

Clogs can be removed by backwashing. Ease of backwashing, given the reactor's position relative to other processes, should be considered when deciding whether to use a downflow (submerged or trickling) or upflow (submerged) filter. For example, Gálvez *et al.* (2003) selected a downflow configuration because the filter was upstream of an anoxic denitrifying filter. Clogging would occur near the top of the aerobic reactor, so backwashing, conducted daily, would be upward, away from the anoxic column (Gálvez *et al.* 2003).

2.4.6 Packing Media

Packing media should be selected to optimize the often-competing requirements of high porosity and large total surface area (Ball 1998). Sufficient porosity and large enough pore spaces are required to promote even aeration throughout the filter, minimize clogging, and provide space for the biofilm to grow (Ball 1998). The aggregate should also be uniform, with minimal fines, to prevent short-circuiting. Ball (1998) recommends that the media contain no more than 4% fines (i.e. – material that will pass through a #100 sieve).

Porosity is favoured over surface area in cases where most of the treatment is due not to the biofilm but to the suspended microbial activity in the working volume. For example, Dermou and Vayenas (2007) noted that there was negligible biosorption of chromium (VI) in trickling filters using plastic and calcitic gravel media; therefore, concentration reduction occurred by the enzymatic action of suspended bacteria. The gravel media had twice the surface area and half the porosity of the plastic media (1059 m^2 vs. 500 m^2 , 0.4 vs. 0.8) (Dermou and Vayenas 2007). The greater working volume of the plastic media allowed more chromium to be treated per unit of reactor volume (Dermou and Vayenas 2007). In addition, the larger pore spaces reduced clogging, allowed for thicker biofilm growth, and allowed more complete use of the reactor volume for greater precipitation of the chromate (III) sediment produced by the enzymatic reaction (Dermou and Vayenas 2007).

Plastic aggregates have become popular in TFs for sanitary wastewater treatment, because of the many shapes and surface treatments available to maximize both porosity and surface area. Plastic media include corrugated cylinders (Knox 1985), plastic spheres (Rowe *et al.* 2000; Yang 2002), and hollow tubes (Dermou and Vayenas 2007), among others. Research has also been

conducted into the use of specialized materials such as expanded aggregate and textile media (Ball 1998), as well as reclaimed waste materials. Table 2-2 lists examples of reclaimed materials that have been studied.

Table 2-2. Reclaimed materials under study for biofiltration of wastewaters

Material	Application	References
Stabilized waste	Nitrogen removal from landfill leachate	(Clabaugh 2001; Jokela <i>et al.</i> 2002)
Compost	Nitrogen removal from landfill leachate	(Jokela <i>et al.</i> 2002)
Crushed limestone	Sanitary wastewater treatment	(Ball 1998)
Crushed glass	Sanitary wastewater treatment; landfill permeable blanket	(Ball 1998; Haydar and Khire 2007; Horan and Lowe 2007)
Tire chips	Nitrogen removal from landfill leachate; septic field; landfill leachate treatment; landfill drainage layer	(Clabaugh 2001; Grimes <i>et al.</i> 2003; Mondal and Warith 2008; Rowe and McIsaac 2005)
Crushed brick	Nitrogen removal from landfill leachate	(Jokela <i>et al.</i> 2002)
Clay shists	Nitrogen removal from sanitary wastewater	(Gálvez <i>et al.</i> 2003)
Slag	Sanitary wastewater treatment	(Ball 1998)
Wood chips	Nitrogen removal from landfill leachate	(Clabaugh 2001)

2.5 Tire Chips and MBG as Filter Media

In Chapter 1, the potential benefits to Ontario of developing new applications for tire chips and MBG were discussed. The suitability of these materials as trickling filter media will be examined in this section.

2.5.1 Tire Chips: Current Uses in Related Applications

Tire chips have been used in landfills in several instances as a leachate drainage layer aggregate in place of gravel. For example, in 2006 approximately 1 million scrap tires from Missouri were shredded and used in a landfill leachate collection layer (Missouri 2006). These tires were removed from what was at the time Missouri's largest remaining tire stockpile by the landowner, TRI-Rinse, who was obligated by state law to eliminate the stockpile (Missouri 2006). Many jurisdictions now have guidelines and regulations in place for the use of tire chips in landfill

liners. These specifications focus on measures to prevent clogging of the drainage layer and tearing of the liner membrane by the steel belting embedded in the tire chips. For example, Arkansas recommends a minimum 6" sand layer between the liner membrane and the tire chip drainage layer if the chips are not metal-free (Arkansas 2003). In addition, a filter layer is required above the tire chips to prevent clogging (Arkansas 2003). Similarly, the state of Nebraska recommends 6" of tire chips laid over 6" of aggregate above the liner (NSRA 2001). Phaneuf and Glander (2003) list several landfills using tire chips in different roles, noting that the state of New York recommends laying 9-12" of aggregate beneath a tire chip drainage layer. The state of Mississippi recommends 12" of tire chips over 12" of sand or other aggregate (Mississippi 2002). It should be noted that no laboratory testing has yet confirmed that wire from tire chips will tear a landfill liner (Edil *et al.* 2004).

In another example of tire chips used in landfills, a study by GeoSyntec Consultants Inc. for the California Integrated Waste Management Board (CIWMB) examined the use of tire shreds for alternative daily cover material, leachate drainage material, final cover foundation layer, protective layer material and as a landfill gas collection layer material (RRI 2002). In Alberta, tire shreds are being used as the lower layer of covering for a pilot compost-over-landfill project designed to reduce methane off-gassing via biofiltration (Edmonton 2007). An important step in developing a more universal approach to the use of scrap tires was the development and publication of an ASTM standard in 2004, ASTM D6270 - 98(2004) "Standard Practice for Use of Scrap Tires in Civil Engineering Applications" (ASTM 2009). This standard covers a wide range of civil applications, including use in highway construction and in drainage layers.

The sandy soils and lack of rock quarries in parts of the southern US provide favourable market conditions for using tire chips as aggregate there. For example, chips have been used as septic drainfield aggregate in South Carolina since 1993, and North Carolina produced a white paper with recommendations for using them in the same application (Grimes *et al.* 2003).

2.5.2 Tire Chips: Physical Characterization and Suitability

Table 2-3 shows the typical composition of a tire, aside from the belting (fabric or steel) used to increase strength. Organo-sulfur compounds are used as vulcanizing agents to catalyze the cross-linking of the natural or synthetic rubber polymers (Amari *et al.* 1999). Cross-linking makes the tire quite resistant to chemical degradation, an important characteristic for a biofilter aggregate.

Table 2-3. Material breakdown by mass of typical passenger tire (Amari *et al.* 1999).

Material (excluding reinforcing fibres)	%
Rubber	62.1
Carbon black	31.0
Extender oil	1.9
Organo-sulfur compounds	1.1
Zinc oxide	1.9
Stearic acid	1.9

Grimes *et al.* (2003) discussed the difference between tire chips and tire shreds. Chips generally pass through a 50.8-mm or smaller sieve and have most of their wire removed, while shreds are less uniform, with sizes generally ranging between 50 mm and 305 mm (Grimes *et al.* 2003). The term "tire chip" also implies control of wire protrusion and size (Grimes *et al.* 2003). Therefore chips rather than shreds are typically specified for landfill and septic applications.

Tire chips have been studied for physical variables that could affect their ability to treat leachate. Research has shown that tire compression under simulated waste loadings does not reduce porosity below acceptable limits (Warith and Rao 2004; Warith *et al.* 2005; Warith and Rao 2006). However Rowe and McIsaac (2005) did conclude that tire chips are less porous than gravel when compressed under waste loading. Tire chip media used in a simulated drainage layer (anaerobic conditions) became clogged within one year, compared to 2 years for gravel (Rowe and McIsaac 2005).

Other research has shown that tire chips do not contribute to leachate toxicity or degrade over time. For example, Grimes *et al.* (2003) found no degradation of or damage to tire chips after 8

years of service as a replacement aggregate for gravel in septic drainfield trenches. The chips were found to increase drainfield storage capacity by 30% due to increased void space (62% for tire chips vs. 44% for gravel) (Grimes *et al.* 2003). Interestingly, Shin *et al.* (1999) estimated individual tire chip surface area to be 6.4 times that of granite and 3.5 times that of briquette ashes. Therefore tire chips would seem to offer both high porosity and high specific surface area. Grimes *et al.* (2003) also noted that they were able to spread each ton of tire chips three times as far as a ton of gravel (21 ft/ton vs. 7 ft/ton), and the chips were much less dense than gravel (800 lbs/cu.yd vs. 2,800 lbs/cu.yd).

There are extra precautions that have to be taken when handling tire chips. As much as possible, steel wire should be removed. For example, LaFlèche Environmental requires that protruding metal fragments make up less than 1% of the mass of chips used in the drainage layer of its bioreactor landfill (LaFlèche 1999). Protruding wires can present a safety problem when handling the chips, and corrosion of the wires discolours effluent water (Grimes *et al.* 2003). Another handling issue that Grimes *et al.* (2003) identified was the “bounciness” of tire chips, which makes them difficult to walk on when they are being laid out. Maintenance-related issues include the importance of screening effluent pipes leading to downstream processes (e.g. – a septic field or reed bed) to prevent clogging by stray tire chips, and the fact that tire drainage layers were found to settle somewhat after the first year of installation (Grimes *et al.* 2003).

Availability of tire chips also needs to be addressed in Ontario regarding their use for leachate treatment. Moose Creek Tire Recycling (MCTR) is currently the only tire chipping facility in eastern Ontario (LeDuc 2008.). The company’s primary customer is the Laflèche landfill, which is located nearby (LeDuc 2008.). Transport from MCTR to other regions could be costly.

2.5.3 Tire Chips: Treatment Performance

Treatment performance of a given packing media depends on several variables inherent to the media, such as its ability to support biofilm growth (related to roughness and specific surface area), its resistance to clogging (pore size and hydraulic conductivity), how well it distributes influent (pore uniformity), whether it is inhibitory to growth, and its abiotic sorption capacity. Several of these variables have been examined for tire chips and presented in the literature.

Shin *et al.* (1999) measured biofilm growth on tire chips during a biomass attachment test. A biofilm established itself after 14 days, with 3.16 and 3.72 mg_{VSS}/cm² after 14 and 31 days, respectively (Shin *et al.* 1999). No significant inhibitory effect from the tire chips was detected, and sequential anaerobic-aerobic biofilm reactors using tire chips effectively degraded dichlorophenol (100% removal in the anaerobic reactor) and chlorophenol (70% removal in the aerobic reactor) (Shin *et al.* 1999).

Tire chips have also been evaluated for removal of more common contaminants, such as ammonia and carbonaceous organic matter. High ammonia levels develop during leachate recirculation, either within treatment systems or within both leachate recirculating and bioreactor landfills (Clabaugh 2001; Del Pozo and Diez 2003). Using small downflow reactors, Clabaugh (2001) tested the ammonia and COD removal performance of tire chips, wood chips, synthetic plastic, and stable refuse. Wood chips were the most effective media for ammonia removal; however, unlike tire chips, wood chips degrade over time, and were not as effective at COD removal (Clabaugh 2001). Mondal and Warith (2008) compared the performance of crumb rubber and tire chips in pilot-scale trickling filters, finding excellent rates of BOD₅, COD, and NH₃-N removal. However, periodic sloughing and clogging did cause the organic matter concentration in the effluent to increase (Mondal and Warith 2008). In general, crumbs were less prone to clogging (Mondal and Warith 2008). Observing a 14-day biofilm growth and sloughing cycle, Mondal and Warith (2008) noted that clogging could be minimized by regular backwashing and a gravel underdrain system. To optimize treatment performance in the case of high organic content, Mondal and Warith (2008) recommended continuous aeration and leachate recirculation combined with a low-to-medium flow rate.

Aside from providing an attachment medium for biofilm, tire chips have some capacity for physical treatment of contaminants. Kim *et al.* (1997) conducted a small batch reactor study testing the ability of ground tire rubber, tire chips, and carbon black to adsorb seven kinds of volatile organic compound (VOC), such as chloroform and ethylbenzene. Kim *et al.* (1997) found that tire chips showed between 46% and 91% less sorption capacity than ground rubber, and that carbon black showed little sorption capacity. The sorption capacities were not significantly affected by changes in pH or temperature, by the presence of other organic

compounds, or by the presence of inorganic compounds (Kim *et al.* 1997). Kim *et al.* (1997) concluded that the primary mechanism for sorbing organic compounds onto tires was sorption onto tire rubber polymers and diffusion through the rubber matrix. Edil *et al.* (2004) extended the research conducted by Kim *et al.* (1997) to a study of tire chips in simulated and actual field conditions. Both the tank- and field-tests demonstrated better sorption capacity of organic compounds by tire shreds than by gravel (Edil *et al.* 2004). Noting that ground rubber would offer further sorptive capacity but reduced hydraulic conductivity and increased cost, Edil *et al.* (2004) concluded that ground tires might serve well as a reactive layer beneath a tire chip bed. While Edil and Kim concluded that tire chips did provide limited abiotic sorption of VOCs, Shin *et al.* (1999) found minimal abiotic sorption of chlorinated phenols, specifically dichlorophenol (DCP) and chlorophenol (CP). However, the biofilms that grew on the chips were effective at removing these chemicals (Shin *et al.* 1999).

2.5.4 MBG: Current Uses in Related Applications

There is almost no research available on the use of unprocessed MBG as a filter medium. However, crushed and graded glass has been used successfully in the state of Washington as a replacement for sand in septic filters (CWC 1997). Mixed glass crushed and graded to ASTM C-33 specifications was tested for two years in a side-by-side comparison to sand as a treatment medium for wastewater from a single-family home (CWC 1997). The graded glass performed as well as the sand, and the state's On-Site Wastewater Technical Review Committee approved it for use in sand filters (CWC 1997). Sand filters using crushed glass as the media have also been successfully used for tertiary treatment at a wastewater treatment plant serving the town of Oswego, New York (Elliott 2001). Located downstream of septic tanks that provide secondary treatment, the filters are 30" thick, subject to an average HLR of $3.7 \text{ gal}\cdot\text{sq}\cdot\text{ft}^{-1}\cdot\text{day}^{-1}$ and provided with recirculation adjustable between 0:1 and 5:1 ratios (Elliott 2001). The maximum glass grain size is $3/8$ " (Elliott 2001). During a comprehensive test period, TSS removal for the treatment plant's septic tanks was 60%, while the addition of the filters raised that value to 98%. At an average effluent DO concentration of 5.0 mg/L, most BOD removal occurred in the filters (in fact, in some cases septic tank BOD effluent concentrations were higher than the influent concentrations) for an overall removal of 96% (Elliott 2001). Ammonia was reduced through

nitrification by 94%, with an associated pH reduction from 7.2 to 6.6 (Elliott 2001). In addition, acid solubility was lower, while permeability and durability (determined via magnesium sulfate soundness testing) were higher for the glass than for natural sand (Elliott 2001).

Horan and Lowe (2007) also studied the use of crushed glass as a trickling filter medium for the tertiary treatment of sanitary wastewater. Three sizes of glass - coarse, medium, and fine - were compared for their ability to remove TSS. The size ranges were 1.5 – 2.5 mm, 0.5 – 1.45 mm, and 0.2 – 1 mm for coarse, medium, and fine glasses, respectively (Horan and Lowe 2007). The fine media produced the cleanest effluent but blinded rapidly, while the coarse media permitted a much higher loading rate but produced a poorer quality effluent (Horan and Lowe 2007).

Medium glass was selected as the best grade (Horan and Lowe 2007). In side-by-side in-situ testing at the Malton (UK) wastewater treatment plant, both medium glass and sand removed 75% of TSS up to an influent concentration of 70 mg/L, but the glass treated 8 – 10% more effluent than the sand during each service/backwashing cycle (Horan and Lowe 2007).

Crushed glass has been used in some landfill applications. Madison County, New York uses crushed glass as drainage layer aggregate in its landfill (Zecca 2006). The county saved \$65,000 (US) by substituting crushed glass for drainage stone in a 1-ft drainage layer over the liner in two waste cells (3.7 acres) (Zecca 2006). The county found the glass to have equal permeability to that of drainage stone (Zecca 2006). In Ohio, Schmucker and Buffalini (2008) conducted laboratory and field evaluations of pulverized MBG as a leachate drainage layer, the results of which satisfied the Ohio Environmental Protection Agency that glass was a suitable aggregate substitute for the application. This finding was based on the material's permeability, lack of clog-inducing carbonate, durability, lack of fines, and workability (i.e. - how easily it can be spread), combined with the fact that it did not damage the underlying geosynthetic landfill liner (Schmucker and Buffalini 1995). As a final example, Haydar and Khire (2007) used crushed glass in field tests on landfill blankets, a new approach to subsurface leachate recirculation.

2.5.5 MBG: Physical Characterization and Suitability

CWC (1997) found that its crushed glass filters were 9 times more porous than sand. Given that the two materials were graded similarly, this greater permeability was attributed largely to the

presence of clay particles that adhere to the sand, and to the cubical shape of the crushed glass granules, which affords more interstitial space than does the spherical shape of the sand granules (CWC 1997).

2.5.6 MBG: Treatment Performance

There is little research into the suitability of glass for aerobic leachate treatment. Glass beads have been used as a packing media in tests using laboratory-scale biofilms. For example, Edgehill (1996) studied the ability of arthrobacter microorganisms inoculated on glass beads in a trickling filter to remove pentachlorophenol (PCP), a persistent organic pollutant (POP) commonly found in wood preservative. Edgehill (1996) noted that arthrobacter adhered to the glass through the production of extra-cellular polymer. Over 300 days, PCP was supplied to the filter as the dominant carbon source in concentrations ranging from 12 – 366 mg/L. Ammonia was also supplied to study its inhibitory affect (Edgehill 1996). Edgehill (1996) found that ammonia did inhibit PCP treatment, including the bacteria's ability to respond to shock loadings; replacement of ammonia with nitrate restored performance. According to Edgehill (1996) the biofilm appeared able to adapt to repeated shock loadings, as effluent PCP concentrations increased less and less after each spike in influent concentration. This resistance could represent an advantage of attached-growth filters over suspended reactors; bacteria that are immobilized and held together by extracellular polymers could be less sensitive than dispersed suspended cells to abrupt substrate changes (Edgehill 1996). In a similar experiment, Soares *et al.* (2003) found that an aerated, submerged, upflow filter inoculated with *Sphingomonas* sp. successfully degraded nonylphenol, another POP.

CHAPTER 3: Materials and Methods

This research was performed to evaluate the suitability of tire chips and mixed broken glass (MBG) as substitutes for conventional aggregate as packing media for aerobic fixed-film treatment of landfill leachate. Each of the three media was packed into its own reactor. The three reactors were then fed from the same initial leachate source and operated simultaneously during each of three experimental runs, hereafter referred to as Phases 1, 2, and 3. The phases varied in terms of aeration method, leachate source, bacterial seeding, hydraulic loading rate, and variables measured in the samples taken from each reactor. The objective of Phase 1 was to confirm whether the system (reactor, media, leachate, and aerator) could provide sufficient nutrients and oxygen to support the biological activity necessary to reduce the organic constituents of the leachate. Based on the findings of this first test, for Phase 2 the HLR was increased and biological activity was stimulated by seeding the leachate with activated sludge from the Ashbridge's Bay sanitary wastewater treatment plant (WWTP) in Toronto. The reactor design, media, and leachate source were modified as discussed later in this chapter. A heterotrophic plate count was added as an additional variable to provide a clearer picture of the biological processes taking place within the leachate. For Phase 3 it was desired to further increase the organic loading rate from Phase 2. Therefore, a synthetic mixture was incorporated into the leachate; this is defined in Section 3.2.

3.1 Packing Media Preparation and Characterization

Tire chips were collected during the spring of 2008 from MCTR. The tire chips were washed and air-dried prior to use in the biofilters. The MBG was collected from the Dufferin Transfer Station in Toronto, and included considerable contamination from other materials, including plastics and organic waste. In order to evaluate the performance of the glass alone, these materials were burned off in a muffle furnace at 550°C. The glass was then washed to remove ash and other residue, and oven-dried at 105°C. The conventional aggregate, river-rock gravel in Phase 1 and crushed rock in Phases 2 and 3, was also washed and oven-dried at 105°C. The crushed rock was used in place of the river-rock gravel after Phase 1 because it was hypothesized that the rougher surfaces of the crushed rock would better retain biofilm than would those of the river rock.

Table 3-1 presents the bulk density (BD), specific gravity (SG), particle density (PD), and porosity measurements for each of the media. The bulk density values found for tire chips and crushed rock agree with the statement made in Section 1.3 that an aggregate bed of tire chips is roughly $1/3^{\text{rd}}$ as dense as one of crushed rock. Tire chips were found to be the most porous media, and river-rock gravel the least porous.

Table 3-1. Physical properties of media.

Property	RRG	MBG	TCH	CRR
Bulk Density (g/mL)	1.73	1.35	0.57	1.55
Specific Gravity	2.79	2.56	1.27	2.69
Particle Density (g/mL)	2.81	2.58	1.27	2.71
Porosity	38%	48%	55%	43%

The particle size distribution for each media was determined before and after the test phases using sieve analysis, to see whether any particle breakdown occurred over the course of the experiment. Sieve analysis was conducted according to ASTM standard C136. Figure 3-1 presents the pre-test grain size distribution of the media. Tire chips have both a larger grain size and greater porosity than the other media. However, the shallower slope of the tire chip curve indicates that it is less uniform than the other media, meaning that permeability is not necessarily higher than it would be for the other media. Crushed rock appears to be the most uniform media. One of the limitations of this test is that it only provides gradation in one dimension: the tire chips and MBG are generally more plate-like than the crushed rock and therefore inherently much less isotropic. While it could be assumed that a plate-like grain structure wherein the plates stack flat on top of each other would reduce porosity, this is not the case for either MBG or tire chips. The large size, acute angles, and protruding wires of the tire chips result in large pores that contribute to high porosity. In addition, the random orientation of both tire chips and MBG as they were poured into place prevented much plate-on-plate stacking. The river-rock gravel was the least porous of the media tested, although it generally had larger particles than either the MBG or crushed rock. The river-rock gravel has more rounded faces than those of the other media; these may allow for tighter packing in the bed.

Figure 3-1. Grain size distribution of media.

3.2 Leachate Source

Knox (1985) noted that laboratory studies conducted at the time tended to use very high-strength leachates, and that often leachate treatment plants were designed to treat leachate strengths much higher than what were actually measured during operation. More recent laboratory studies (e.g. - Rowe and McIsaac 2005) have used leachate collected directly from active landfills in order to best approximate in-situ conditions, despite the compositional variations inherent to real leachate. However, the leachates from such studies have come from large active landfills, and as such are at or near their peak contaminant strengths. Considering that one of the major criticisms of the use of aerobic biofilters (i.e. - trickling filters) for leachate treatment is their tendency to clog when treating high-strength leachates, and considering the fact that a simple, low-cost treatment system would best benefit small communities operating older, more stabilized landfills, the source of leachate selected for Phase 1 was from a closed site, the Keele Valley Landfill (KVL), located in Vaughan, Ontario. Prior to its closure, KVL received municipal solid waste from the Greater Toronto Area. Rowe (1995) provides a detailed characterization of the landfill when it was in operation. The leachate was collected from the sedimentation chamber at the outlet of the leachate collection system. All leachate collected from the landfill enters this chamber, where it mixes with groundwater from beneath a nearby former unlined dump prior to

being discharged to the public sanitary sewer system. Leachate from the same collection point was used in previous studies (e.g. - Mondal and Warith 2008; Rowe and McIsaac 2005). Interestingly, sampling of the KVL leachate collected for this experiment demonstrated that it had lost much of its strength since it had been used in the above-mentioned studies. The sample point was the same for all studies, and KVL confirmed that no significant changes to the leachate handling system or waste pile had occurred in the intervening time (Benda 2008 May 13). Therefore the change in strength is likely due to increased stabilization of the waste pile over time. This more stabilized leachate was considered a good representative of the leachates that might be found in older, rural landfills.

For Phase 2, KVL leachate was mixed with leachate from the Glanbrook Landfill in Hamilton, as described in further detail in Section 3.5. This is an active municipal landfill; however, it had primarily handled ash from the city's incinerator. The incinerator had recently been closed, and as of autumn 2008 Glanbrook began receiving municipal waste directly. This presented an opportunity to study a relatively low-strength leachate from an active site.

For Phase 3, chemicals were added to provide more organic carbon, as described in further detail in Section 3.6. These are listed in

Table 3-2. The mixture was based on a synthetic leachate used by Mondal and Warith (2008) and originally designed by Hrapovic (2001) to simulate KVL leachate as sampled when the landfill was still in operation. The base leachate, used as a source of conservative ions, was the 50% KVL and 50% HGL mixture used in Phase 2.

Bacterial seed was added to the leachate in Phase 2 and Phase 3 (Sections 3.5 and 3.6, respectively). The seed was sludge obtained from the recirculation line of the activated sludge reactor at the Ashbridge's Bay plant. The seed was stored at 4°C and was not subjected to any deliberate acclimation process. Rather, the ability of the bacteria to adjust to the leachate source would be demonstrated by the data collected during the experimental runs.

Table 3-2. Phase 3 leachate constituents.

Constituent ⁹	Quantity (per L)
Acetic Acid (CH ₃ COOH)	4.7 mL
Propionic Acid (CH ₃ CH ₂ COOH)	1.1 mL
n-Butyric Acid (CH ₃ CH ₂ CH ₂ COOH)	0.7 mL
Potassium Phosphate (K ₂ HPO ₄)	15 mg
Potassium Bicarbonate (KHCO ₃)	166 mg
Potassium Carbonate (K ₂ CO ₃)	162 mg
Sodium Nitrate (NaNO ₃)	25 mg
Sodium Bicarbonate (NaHCO ₃)	1506 mg
Calcium Chloride (CaCl ₂)	1441 mg
Magnesium Chloride Hexahydrate (MgCl ₂ ·6H ₂ O)	1557 mg
Magnesium Sulfate (MgSO ₄)	78 mg
Ammonium Bicarbonate (NH ₄ HCO ₃)	1220 mg
Urea (CO(NH ₂) ₂)	348 mg
Sodium Hydroxide (NaOH)	To titrate to pH 7
Base Leachate	To make 1 L

3.3 Reactor and Circuit

A schematic of the reactor circuit used in Phase 1 is shown in Figure 3-2. The circuit components are detailed in Table 3-3. The packing media were poured into the cylinders to create a randomly packed orientation. The tire chips were further rammed into the cylinder using a metal rod in order to minimize major void spaces. The packing media were completely submerged in order to promote uniform distribution of substrate and oxygen, and to provide maximum nutrient loading to the biofilm. To facilitate media saturation and flow rate adjustment, an upflow circulation mode was selected. Upflow operation helps to avoid flow irregularities (e.g. – turbulent flow caused by sudden flow rate changes, such as a clogged pore) that could complicate substrate utilization. A recirculating batch design was chosen instead of a single-pass design in order to approximate complete mixing (Park *et al.* 1982).

⁹ Relative proportions of Acetic, Propionic, and n-Butyric acids vary slightly from figures provided by Mondal (2006) due to availability of chemicals in lab. NaCl, present in Hrapovic's (2001) design, was omitted this time because there was already a considerable presence of Cl⁻ in the leachate, and because Na⁺ would not contribute significantly to precipitation. The trace metal solution used by Hrapovic (2001) was also omitted.

Figure 3-2. Phase 1 reactor circuit schematic.

Table 3-3. Apparatus materials.

No.	Name	Qty.	Description
1	Reactor Vessel	1	6" nom. dia. PVC cylinder
2	Reservoir	1	Phase 1: 121 L PE plastic container Phases 2 & 3: 7 L PE plastic container
3	Circulation Pump	1	Submersible pump, Algreen Nova
4	Oxygen Source	1	Aquarium aerator, Tetra Whisper 10-30 gal.
5	Valves & Fittings	-	PE plastic
6	Collector Funnel	1	PE plastic funnel
7	Tubing	-	Soft clear PVC tubing. 1/2" ID × 5/8" OD for leachate circulation; 3/16" ID × 1/4" OD/ for air supply
8	Clamps & Sealants	-	Stainless steel wormgear-type clamps, hot-melt glue, "Superglue", electrical tape, Silicone sealant

3.3.1 Reactor Column Design

The reactors were constructed from 6" (nominal) PVC pipe. PVC provides excellent chemical and moisture resistance (PacWest 2009). Each reactor was sealed at the bottom with a PVC cap and left open to room conditions at the top. Each reactor had an overall height of 1.00 m, a filter

column height of 0.90 m, and a column internal diameter of 0.152 m. These dimensions fall within the range of similar experiments described in the literature (e.g. - Clabaugh 2001; Rowe and McIsaac 2005). Column diameter was selected to be large enough to avoid possible sidewall effects and small enough to facilitate lateral mixing within the column. In an anaerobic fixed film reactor study, Rowe *et al.* (2002) found no evidence of additional clogging along reactor sidewalls that would indicate preferential flow. Although the grain size/column diameter ratios of the media tested in the present case were relatively large¹⁰ (e.g. - tire chips), it was assumed that flow variations due to the relative non-uniformity of the pore spaces between these aggregates would most likely overshadow any sidewall effects. The working reactor volumes, functions of the media porosities listed in Section 3.1, were $6.2 \times 10^{-3} \text{ m}^3$, $7.8 \times 10^{-3} \text{ m}^3$, $9.0 \times 10^{-3} \text{ m}^3$, and $7.0 \times 10^{-3} \text{ m}^3$ for river-rock gravel, mixed broken glass, tire chips, and crushed rock, respectively.

3.3.2 Other Circuit Components

Each reactor circuit included a submersible pump placed in a reservoir to recirculate the leachate through the reactor. Because the leachate was of low strength, the reservoir was filled with a relatively large quantity of it, 12 L, in order to provide more substrate. Circuit components were connected using transparent flexible vinyl tubing, and flow rate was manually controlled using a ball valve. Room air was supplied to the reservoir using a fish-tank aerator and gate valve. Leachate was aerated in the reservoir rather than in the reactor to promote more uniform gas distribution. The leachate was aerated sufficiently to provide excess oxygen under steady-state operation; therefore, nutrient supply would be the limiting growth factor.

3.3.3 Design Modifications for Phases 2 and 3

In order to reduce the impact of biochemical reactions and physical processes taking place in the reservoirs instead of in the reactors, in Phases 2 and 3 smaller reservoirs were used, and only enough leachate (3 L per reservoir) was maintained in them to submerge each pump. In addition, aeration was provided directly to the bottom of the reactor instead of to the reservoir. Although this had the potential to create preferential air pathways through the filter, results from Phase 1 indicated that it would be possible to maintain excess oxygen throughout the column. A break

¹⁰ The study by Rowe *et al.* (2002) used uniform 6-mm dia. glass beads in a 5-cm dia. reactor.

was made in the leachate return line to ease collection of leachate during flow measurement. The break consisted of a funnel used to catch leachate as it dripped from a downspout at the top of the reactor column. Figure 3-3 shows the revised circuit.

Figure 3-3. Phase 2 and 3 reactor circuit schematic.

3.4 Phase 1: Treatment of Unseeded Aged Leachate

The ability of the bioreactors to establish a biofilm in unaltered aged leachate was evaluated in Phase 1. The three media tested were river-rock gravel from a The Home Depot retail outlet in Toronto, tire chips from Moose Creek Tire Recycling (MCTR) in Moose Creek, Ontario, and mixed broken glass (MBG) from the Dufferin Transfer Station in Toronto. Leachate from the closed Keele Valley Landfill (KVL) in the City of Vaughan was used. As collected from the landfill, the leachate had a negligible dissolved oxygen (DO) concentration. Therefore, circulation through the reactor did not begin until DO had been increased by aeration to 2.0 mg/L. This is the minimum concentration typically required by nitrifying bacteria (Knox 1985). Leachate was circulated through each reactor circuit at a flow rate sufficient to achieve an HLR

of approximately $2 \text{ m}^3 \text{ m}^{-2} \text{ day}^{-1}$, consistent with that of a low-rate trickling filter (Reynolds and Richards 1996). Flow rate was measured daily using a graduated cylinder and stopwatch after Clabaugh (2001). Table 3-4 summarizes the leachate variables measured, the test methods and equipment used, and the frequency of measurement.

Table 3-4. Variables measured during Phase 1.

Variable	Test Method (from Clesceri <i>et al.</i> 1998)	Equipment	Sample Freq.
Temp. (°C)	2550 B	Oakton pH meter temp. probe	1/day
DO (mg/L)	4500-O G	Horiba DO meter	1/day
pH	4500-H ⁺ B	Horiba D-53 pH meter (#6066 probe)	1/day
COD (mg/L)	5220 D (Closed Reflux, Colorimetric)	Photospectrometer	2/wk
TS (mg/L)	2540 B	Oven, scale	start/end
TSS (mg/L)	2540 D	Oven, scale, Whatman Paper	start/end

Leachate samples were collected from the top (outlet) of each reactor and from the reservoir in clean glass preserve jars. COD, Total Solids (TS), and Total Suspended Solids (TSS) were monitored as treatment performance indicators. The run was considered complete once COD had stabilized. Differences between the entrance and exit variable values of the filter column, particularly differences in dissolved oxygen (DO) concentration, served indirectly as indicators of biological activity. Leachate temperature and DO were used to monitor environmental conditions within the reactors. Temperature was measured to ensure there were no significant fluctuations due to changing lab conditions. DO was measured to ensure it remained above the minimum concentration. pH was monitored to ensure it remained between 6.5 and 9, the required range for most biological treatment (Reynolds and Richards 1996).

3.5 Phase 2: Treatment of Aged Leachate with Bacterial Seed Added

Bacterial seed (activated sludge) was added to the leachate in Phase 2 in an effort to improve the performance of the bioreactors. Leachates from KVL and from the City of Hamilton's Glanbrook Landfill (HGL) were used in a well-mixed 1:1 ratio. The HGL leachate had a lower COD and a higher pH than those of the KVL leachate, as shown in Chapter 4, Table 4-1. These sources were mixed in order to provide a more alkaline substrate in an effort to stimulate

precipitation of CaCO_3 (and therefore clogging) without losing too much initial COD. 1 g of bacterial seed was added per L of leachate. The three media evaluated in Phase 2 were crushed rock, tire chips, and MBG. The reactor design was revised as discussed in Sub-Section 3.3.3. Given that bacterial seed was used and COD was lower in this run, it was anticipated that the reactors could support a higher HLR. Therefore, it was doubled to $4 \text{ m}^3 \text{m}^{-2} \text{day}^{-1}$. The flow rate was measured daily. Table 3-5 summarizes the leachate variables measured, the test methods and equipment used, and the frequency of measurement.

Table 3-5. Variables measured during Phase 2.

Variable	Test Method (from Clesceri <i>et al.</i> 1998)	Equipment	Sample Freq.
Temp. ($^{\circ}\text{C}$)	2550 B	Horiba D-53 pH/ion meter (#4163 probe)	1/day
DO (mg/L)	4500-O G	Horiba DO meter	1/day
pH	4500- H^+ B	Horiba D-53 pH meter (#6066 probe)	1/day
Conductivity ($\mu\text{S}/\text{cm}$)	2510 B	Horiba D-53 pH/ion meter	2/wk
HPC (CFU/mL)	9215 C (Spread Plate Method)	Tryptic Soy Broth, Agar	2/wk
COD (mg/L)	5220 D (Closed Reflux, Colorimetric)	Photospectrometer	2/wk
BOD_5 (mg/L)	5210 B	Horiba DO meter	2/wk
$\text{NH}_3\text{-N}$ (mg/L)	4500	Horiba D-53 pH/Ion meter (#5002 probe)	2/wk
TS (mg/L)	2540 B	Oven, scale	2/wk
TSS (mg/L)	2540 D	Oven, scale, Whatman Paper	2/wk

To simplify data analysis, samples were collected from the tops of the reactors only. In addition to COD, TS, and TSS, 5-day biochemical oxygen demand (BOD_5) and nitrogen as ammonia ($\text{NH}_3\text{-N}$) were monitored as performance indicators. No bacterial seed was used in the BOD_5 tests since it was already present in the leachate. The run was considered complete once BOD_5 had stabilized. Conductivity and heterotrophic plate count (HPC) were measured as environmental variables in addition to temperature, DO, and pH. Conductivity was measured in order to quantify the high ionic salt concentrations present in the leachate. Large concentrations can interfere with bacteriological growth and precipitate as clog material in the alkaline conditions of the leachate (Reynolds and Richards 1996). Further, as a check on TSS

measurements, conductivity can be correlated with Total Dissolved Solids (TDS) according to the following equation:

$$\text{TDS} = K \cdot C \quad (3.1)$$

where TDS (the solids portion remaining when TSS is subtracted from TS) is expressed in mg/L, C is conductivity ($\mu\text{S}/\text{cm}$), and K is an empirical constant ranging between 0.55 and 0.9 and dependent on the type of salts present and the temperature (Clesceri *et al.* 1998; Reynolds and Richards 1996). HPC was used to monitor changes in the live bacterial populations present in the leachate.

3.6 Phase 3: Treatment of Synthetic Acetogenic Leachate

The performance of the bioreactors in treating more acetogenic leachate was evaluated in Phase 3. Because an acetogenic landfill source was not readily available at the time of the experiment, the leachate mixture from Phase 2 was used, and a chemical mixture, detailed in Section 3.2, was added to significantly increase the organic carbon and nutrient content available to the microbes. 1 g/L of activated sludge was again added. The same media from Phase 2 were used as well. The HLR was increased to $6 \text{ m}^3 \text{m}^{-2} \text{day}^{-1}$, and the flow rate was measured daily. The same variables were measured as in Phase 2 (Table 3-6).

Table 3-6. Variables measured during Phase 3.

Variable	Test Method (from Clesceri <i>et al.</i> 1998)	Equipment	Sample Freq.
Temp. ($^{\circ}\text{C}$)	2550 B	Oakton pH meter (temp. probe)	1/day
DO (mg/L)	4500-O G	Horiba DO meter	1/day
pH	4500-H ⁺ B	Oakton pH meter	1/day
Conductivity ($\mu\text{S}/\text{cm}$)	2510 B	Horiba D-53 pH/ion meter	2/wk
HPC (CFU/mL)	9215 C (Spread Plate Method)	Tryptic Soy Broth, Agar	2/wk
COD (mg/L)	5220 D (Closed Reflux, Colorimetric)	Photospectrometer	2/wk
BOD ₅ (mg/L)	5210 B	Horiba DO meter	2/wk
NH ₃ -N (mg/L)	4500	Horiba D-53 pH/Ion meter (#5002 probe)	2/wk
TS (mg/L)	2540 B	Oven, scale	2/wk
TSS (mg/L)	2540 D	Oven, scale, Whatman Paper	2/wk

Samples were collected from the tops of the reactors. The run was considered complete once BOD₅ had stabilized.

3.7 Examination of Sediment Deposition

Prior to the reactor tests, a sample set of aggregates from each media type (excluding river-rock gravel) were washed and dried, numbered, weighed, and mixed randomly with the rest of the media as the reactors were packed. After draining the leachate from the reactors at the end of the experiment, these samples were collected and weighed again in order to determine an average sample deposition for each media.

While emptying the reactors, samples of the media were taken from the top, middle, and bottom of each one. Depositions were scraped off and collected from the aggregates in the samples. The depositions were oven-dried at 105°C for 1.5 hours, weighed, oven-heated at 550°C for 45 minutes to burn off Volatile Solids (VS) and leave Fixed Solids (FS) as residue, and weighed again. The purpose of this procedure was to determine how the FS:TS ratio of the material deposited on each media type changed with depth. VS can provide an indication, though not an accurate measure (due to the losses of mineral salts) of organic matter present in the sample.

CHAPTER 4: Results and Discussion

Table 4-1 summarizes the initial values of the variables measured at the beginning of each phase. Each phase consisted of one experimental run. For brevity, the reactors packed with river-rock gravel, crushed rock, MBG, and tire chips will henceforth be referred to as RRG, CRR, MBG, and TCH, respectively.

Table 4-1. Summary of input values, Phases 1 - 3.

Variable	Phase 1	Phase 2	Phase 3
Media Compared	RRG, MBG, TCH	CRR, MBG, TCH	CRR, MBG, TCH
Leachate Composition	100% KVL	50% KVL, 50% HGL	50% KVL, 50% HGL, Additives
Activated Sludge (g/L)	None added	1	1
HLR ($\text{m}^3 \text{m}^{-2} \text{d}^{-1}$) after daily adjustment	2	4	6
Temp. ($^{\circ}\text{C}$)	25.4	25.0	23.8
DO ¹¹ (mg/L)	Min. 2.00	Min. 2.00	Min. 2.00
pH	7.65	8.98	7.53
Conductivity ($\mu\text{S}/\text{cm}$)	Not measured	9,560	12,770
HPC (CFU/mL)	Not measured	1.1×10^6	1.8×10^7
COD (mg/L)	2,443	2,014	5,243
BOD ₅ (mg/L)	Not measured	110	464
NH ₃ -N (mg/L)	Not measured	363	123
TS (mg/L)	9,650	10,244	14,813
TSS (mg/L)	0	280	985

4.1 Phase 1: Treatment of Unseeded Aged Leachate

Figure 4-1 presents the COD over the course of Phase 1. Table 4-2 summarizes the initial and final COD values, final removal percentage, and peak removal percentage.

¹¹ Leachate was aerated to min. 2.00 mg/L immediately prior to the start of each experimental run.

Figure 4-1. Phase 1 COD.

Table 4-2. Phase 1 COD removal.

COD	RRG	MBG	TCH
Initial (mg/L)	2,443	2,443	2,443
Final (mg/L after 21 days)	1,929	1,929	2,386
Final % Removed (after 21 days)	21%	21%	2%
Peak % Removed	21% (Day 21)	35% (Day 11)	37% (Day 6)

From these figures, based on samples taken from the top of the reactor, it is evident that COD removal was limited in the stabilized leachate without bacterial seeding. Given that TSS (Figure 4-5) was 0 or not detectable at the beginning of the run, it is likely that there was neither a sufficient population of microbes present in the leachate to degrade the oxygen demand present, nor sufficient easily biodegradable organic matter present to promote microbial growth. Reduction in oxygen demand may have been solely due to oxidation of inorganic ions, such as iron, and/or precipitation of these ions under the alkaline and now aerobic leachate conditions. COD increased in all reactors after day 11. This is probably related to a drop in temperature and an increase in average flow rate that occurred at the time (Figure 4-2 and Figure 4-3). The increased flow may have resuspended previously settled oxidisable matter, and the drop in

temperature would have slowed countervailing precipitation reactions. Perhaps due to the stabilization of flow rates and increasing temperatures (thus increased precipitation) after Day 15, COD in the MBG and RRG leachates once again declined. The tire chip COD remained high; this is possibly due to continued oxidation of iron that may have detached from the tire wires as surrounding portions rusted. A sharp increase in TSS at the end of the run, which upon observation appeared to consist mostly of rust flakes, supports this argument. In addition, the leachate from the tire chip reactor was less transparent than that of the other reactors due to the accumulation of ferric oxide.

Figure 4-2. Difference between flow rate at start of day (FRmeas) and that after adjustment (FRadj) in RRG reactor. Large initial difference between these values implies lower average flow rate during first days of test.

Figure 4-3. Phase 1 leachate temperatures, showing drop mid-way through run.

Figure 4-4. Phase 1 Total Solids.

Figure 4-5. Phase 1 Total Suspended Solids.

TS increased over the duration of the run almost equally for all media, from 9,650 mg/L to between 11,000 mg/L (TCH) and 11,167 mg/L (MBG). This was counter to the results of later phases, and somewhat counter-intuitive. As noted previously, the leachates from RRG and MBG had no TSS. Therefore the additional TS may have come from soluble solids (small enough to pass through the TSS filter) that remained on the media after washing. This might include very fine clay particles from RRG, ash residue from MBG (after burning of residue), and again ferric oxide from TCH.

4.2 Phase 2: Treatment of Aged Leachate with Bacterial Seed Added

Figure 4-6 presents COD during Phase 2. Table 4-3 summarizes the beginning, ending, and peak values for the run. MBG (30% removal) and TCH (27% removal) performed better than CRR (19%), however overall the COD removal limit for the reactors appears to be approximately 1500 mg/L, given the leachate and bacteria supplied.

Figure 4-6. Phase 2 COD.

Table 4-3. Phase 2 COD removal.

COD	Crushed Rock	MBG	Tire Chips
Initial COD (mg/L)	2,014	2,014	2,014
Final COD (mg/L after 17 days)	1,614	1,414	1,486
Final % Removed (after 17 days)	19%	30%	27%

As illustrated in Figure 4-7 and Table 4-4, all three reactors achieved very high BOD₅ removal rates, with CRR providing the best removal at 91%, although only by a few percent.

Figure 4-7. Phase 2 BOD₅.

Table 4-4. Phase 2 BOD₅ removal.

BOD ₅	Crushed Rock	MBG	Tire Chips
Initial BOD ₅ (mg/L)	110	110	110
Final BOD ₅ (mg/L after 17 days)	10	14	13
Final % Removed (after 17 days)	91%	88%	88%

The BOD:COD ratio (Figure 4-8) was well below 0.10 in all reactors throughout the run, decreasing as COD stabilized and BOD continued to be removed. Therefore, when seeded with bacteria the aerobic fixed-film reactors were able to effectively reduce the biodegradable fraction of the total oxygen demand at very low BOD:COD ratios.

Figure 4-8. Phase 2 BOD:COD ratios.

Ammonia concentrations (Figure 4-9) increased by 1.5 – 4 times their initial value by Day 8. This is presumably due to biological activity in the leachate. The low degradable carbon content of the stabilized leachate achieved after the first week or so of the run would enable nitrifying bacteria to establish themselves, resulting in the removal during the second week of the $\text{NH}_3\text{-N}$ generated in the first, as reflected in Figure 4-9. Since nitrification lowers pH, a decrease in pH would be anticipated over the second week of the run, and does occur, as illustrated in Figure 4-10. However, as summarized in Table 4-5, the reactors did not significantly reduce ammonia beyond the initial concentration, with MBG, CRR, and TCH achieving 38%, 12%, and 9% removal, respectively.

Figure 4-9. Phase 2 $\text{NH}_3\text{-N}$.

Figure 4-10. Phase 2 pH

Table 4-5. Phase 2 NH₃-N.

NH ₃ -N (mg/L)	Crushed Rock	MBG	Tire Chips
Day 3	434	343	311
Final	382	211	284
% Removed	12%	38%	9%

Knox (1985) noted that increased use of leachate recirculation in landfills to promote methanogenic conditions and thereby increase methane production for use as fuel results in leachate with a high concentration of ammonia and low degradable carbon content (i.e. - a low BOD:NH₃-N ratio). This finding is affirmed by the low BOD:NH₃-N ratios found for this stabilized leachate, shown in Figure 4-11. In addition, research has shown that levels of non-biodegradable ("hard") COD are related not to influent (raw leachate) concentrations of organic matter, but to influent concentrations of NH₃-N (IPPC 2007). This correlation is probably due to hard COD being released with ammonia during waste degradation, or to generation of hard COD during nitrification (IPPC 2007).

Figure 4-11. Phase 2 BOD:NH₃-N ratios.

TS and TSS are presented in Figure 4-12, Figure 4-13, and Table 4-6. Some TS removal was achieved, with MBG performing best, followed by CRR and TCH. As TSS was relatively low,

the majority of the TS removed were TDS; this is reflected in the reduction in conductivity shown in Figure 4-14. All three reactors achieved complete or almost complete removal of TSS; again MBG performed the best.

Figure 4-12. Phase 2 Total Solids.

Figure 4-13. Phase 2 Total Suspended Solids.

Table 4-6. Phase 2 TS and TSS.

Solids	Crushed Rock	MBG	Tire Chips
TS Initial (mg/L)	10,244	10,244	10,244
TS Final (mg/L)	8,750	8,250	8,900
TS removal (%)	15%	19%	13%
TSS Initial (mg/L)	280	280	280
TSS Final (mg/L)	20	0	20
TSS removal (%)	93%	100%	93%

Heterotrophic plate count results (Figure 4-15) did not correlate with the other leachate variables. Accurate plate counts were difficult to achieve because of the variety of bacteria present in the leachate. They did confirm that there was biological activity in the leachate throughout the run.

Figure 4-14. Phase 2 Conductivity.

Figure 4-15. Phase 2 Heterotrophic plate count.

4.3 Phase 3: Treatment of Synthetic Acetogenic Leachate

Figure 4-16 presents the COD over the course of Phase 3 and Table 4-7 summarizes the initial and final data. Approximately 40 – 50% of COD removal was achieved by Day 10, with only another 5 – 15% removed by Day 21. This is not surprising, given that BOD₅ removal (Figure 4-17 and Table 4-8) was almost complete by Day 10, and that therefore the BOD:COD ratio (Figure 4-18), low to begin with, dropped to extremely low levels from Day 10 onward. All three media performed similarly for both COD and BOD₅ removal.

Figure 4-16. Phase 3 COD.

Table 4-7. Phase 3 COD removal.

COD	Crushed Rock	MBG	Tire Chips
Initial COD (mg/L)	5,243	5,243	5,243
Final COD (mg/L after 21 days)	2,343	2,357	2,329
Final % Removed (after 21 days)	55%	55%	56%

Figure 4-17. Phase 3 BOD₅.

Table 4-8. Phase 3 BOD₅ removal.

BOD ₅	Crushed Rock	MBG	Tire Chips
Initial BOD ₅ (mg/L)	464	464	464
Final BOD ₅ (mg/L after 17 days)	8	4	5
Final % Removed (after 17 days)	98%	99%	99%

Figure 4-18. Phase 3 BOD:COD ratios.

Ammonia (Figure 4-19 and Table 4-9) exhibited the same initial peak followed by a steady (approximately linear) decline as in Phase 2. Removal was fairly significant, with 97% for CRR, 91% for MBG, and 67% for TCH. Again a slight pH decrease was evident in the last days of the run (Figure 4-20). Sufficient ammonia was removed that the BOD:NH₃-N curve (Figure 4-21) began to take a concave form, with the bottom occurring around Day 10.

Figure 4-19. Phase 3 $\text{NH}_3\text{-N}$.

Table 4-9. Phase 3 $\text{NH}_3\text{-N}$ removal.

$\text{NH}_3\text{-N}$ (mg/L)	Crushed Rock	MBG	Tire Chips
Initial	504	504	504
Final	16	44	169
% Removed	97%	91%	67%

Figure 4-20. Phase 3 pH.

Figure 4-21. Phase 3 BOD:NH₃-N ratios.

TS and TSS are presented in Figure 4-22, Figure 4-23, and Table 4-10. TS removal was limited, with the three media performing within a few percentage points of each other (MBG was best).

TSS removal was almost complete however, with 98% removal for CRR, 92% for MBG, and 90% for TCH. Conductivity (Figure 4-24) decreased steadily for all media.

Figure 4-22. Phase 3 Total Solids.

Figure 4-23. Phase 3 Total Suspended Solids.

Table 4-10. Phase 3 TS and TSS.

Total Solids	Crushed Rock	MBG	Tire Chips
TS Initial (mg/L)	14,813	14,813	14,813
TS Final (mg/L)	12,400	11,963	12,263
TS removal (%)	16%	19%	17%
TSS Initial (mg/L)	985	985	985
TSS Final (mg/L)	20	80	100
TSS removal (%)	98%	92%	90%

Figure 4-24. Phase 3 Conductivity.

As in Phase 2, the heterotrophic plate count for Phase 3 appeared to demonstrate that bacterial populations remained relatively stable throughout the run. This indicates that sustainable populations can be achieved in leachate when using sanitary wastewater sludge as seed. The lack of COD removal in Phase 1 underlines the importance of having a viable bacterial population present during reactor start-up.

Figure 4-25. Phase 3 heterotrophic plate count.

4.4 Post-Test Examination

One possible concern with glass is that it could break into smaller pieces over time; these smaller grains could then shift into existing pore spaces and contribute to clogging of the filter.

Therefore a sieve test was repeated after completion of Phase 3, using the media recovered from the reactors after they had been thoroughly washed and air-dried. As Figure 4-26 through Figure 4-28 indicate, none of the media exhibited much change in size distribution outside of what might be expected due to test error. However, as Table 4-11 shows, gradation at the end of testing revealed that tire chips produced the most fines in terms of percent of total media mass, while crushed rock had the least. These fines, considered here to be particles passing through a #10 (2.00 mm) sieve as per Eash *et al.* (2008), could contribute to more rapid clogging.

Figure 4-26. Pre- vs. post-test grain size distribution of crushed rock.

Figure 4-27. Pre- vs. post-test grain size distribution of MBG.

Figure 4-28. Pre- vs. post-test grain size distribution of tire chips.

Table 4-11. Post-test fines in packing media.

Sieve #	Size (μm)	Crushed Rock	MBG	Tire Chips
#10 > x > #20	2000 > x > 841	0.773 g	13.970 g	6.904 g
#20 > x > #40	841 > x > 420	0.381 g	5.689 g	5.903 g
#40 > x > #200	420 > x > 74	1.031 g	8.547 g	6.860 g
#200 > x	74 > x	0.031 g	0.089 g	0.246 g
Fines as % total media mass		0.019%	0.147%	0.245%

Table 4-12 presents the average aggregate deposition masses and standard deviations. On average, the net deposition on crushed rock and MBG was minimal (indeed slightly negative), while tire chips did have measureable accumulation. These findings are consistent with visual observations of the media; the tire chips had visible orange/rust-coloured sediment, while the other two media appeared relatively unchanged.

Table 4-12. Sediment deposition on aggregate samples.

Deposition Mass (g)	Crushed Rock	MBG	Tire Chips
Average	-0.002	-0.02	0.26
Std. Dev.	0.007	0.04	0.19

Figure 4-29 presents the FS:TS ratios of deposition on the media at three different sections of each reactor column: bottom (inlet), middle, and top (outlet). The top samples of crushed rock and MBG media appeared to have less overall deposition than those of the middle and bottom samples (the bottom samples had the most). Deposition appeared to be more evenly distributed through the three tire chip sample depths. The tire chips had the lowest average FS:TS ratio (71.3%), followed by MBG (83.2%) and crushed rock (92.1%). This indicates the presence of more oxidizable material in the tire chip deposition. There did not appear to be much correlation between sample locations within the column (i.e. – Top/Middle/Bottom) and FS:TS ratio.

Figure 4-29. FS:TS ratios of deposition.

CHAPTER 5: Conclusions

Given that little deposition could be detected on any of the media by the end of Phase 3, and that the higher mass and VS percentage of the deposition on tire chips did not seem to translate into a performance advantage, it appears that most treatment occurred in the suspended phase.

Therefore, under both stabilized and moderate-strength leachate conditions, the packing media served mainly as a physical filter and as a means of mixing the leachate while maintaining laminar flow. In such a role, tire chips and MBG both appear to be suitable substitutes for crushed rock. Treatment performance was similar during all three phases, and no appreciable breakdown of MBG (or tire material) into finer particles was detected over the 60-day duration of the experiment. The limited deposition prevented a conclusive determination as to whether the greater proportion of fines in tire chips and MBG compared to crushed rock would lead to faster clogging in those media.

Significant COD removal from aged leachate did not occur in the bioreactors without bacterial seeding (Phase 1). Once seeded (Phase 2), the bioreactors were able to effectively remove almost 100% of BOD₅ in stabilized leachate at BOD₅:COD ratios less than 0.1. However, the lowest COD achievable was 1,414 mg/L for MBG, 1,486 mg/L for tire chips, and 1,614 mg/L for crushed rock under the same conditions. Under more acetogenic conditions (Phase 3), crushed rock, MBG, and tire chips exhibited similar removal performance of COD (55 – 56%) and BOD₅ (98 – 99%). Most removal occurred within the first ten days of the run.

River-rock gravel, MBG, and tire chips may initially contribute to increased TS concentrations until they have been scoured sufficiently by leachate flow to remove persistent residues. In addition, while able to remove the ammonia generated during carbon degradation, the seeded reactors were unable to nitrify NH₃-N much below the concentration present in the aged leachate. MBG provided the best performance, with 38% removal. However, high ammonia removal rates were achieved for all three media under moderate organic loading (Phase 3), with crushed rock performing best (97%), followed by MBG (91%) and tire chips (67%). Finally, bacterial seed from sanitary wastewater activated sludge maintained steady viable populations without any acclimation period when exposed to both aged and more acetogenic leachates.

CHAPTER 6: Recommendations

Precise flow rate control and mass balance determination were not possible in this experiment and therefore outside its scope. The use of a single three-channel positive-displacement (peristaltic) pump would enable precise control of flow rate, and therefore calculation of, for example, mass of COD treated over time. The addition of gas collection bags together with gas analysis by chromatography or mass spectrometry would allow calculation of the gaseous mass flow out of the system, including evaporation. Aeration rate should also be precisely controllable and measurable. Control of liquid and air flow in, and gas flow out would allow for estimation of the quantity of solids generated in each reactor by precipitation and metabolic activity. Comparison of solids generated for each media could then allow estimation of which media would be most resistant to clogging in the field. A comparison test between one of the fixed-film bioreactors and a bioreactor supplied only with an equivalent working volume of aerated leachate (no packing media) would enable estimation of the impact on treatment of the packing media. Complete mixing would be required in this "empty" reactor.

A report co-published by the environmental agencies of England, Scotland, Northern Ireland, and Wales providing guidelines for landfill technology selection stated that the BOD₅ test will under-report the extent to which organic compounds can be degraded by an acclimatized bacterial population in an aerobic biological treatment plant (IPPC 2007). The report noted that the BOD₅ test uses a standard sewage-based bacterial seed; therefore the 5-day incubation period might not provide sufficient time for acclimatization (IPPC 2007). This effect would have a greater impact on raw leachate (influent) measurements than on already low plant effluent values, and therefore the reported difference between influent and effluent values (the mass removed) would be too low. Because most successful aerobic leachate treatment plants remove a far greater mass of organic compounds than predicted by even a 20-day BOD test, the report recommended that COD values should be used for design purposes (IPPC 2007). While BOD seeding was not used in this test, omission of the BOD test, or limitation of its use to the beginning and end of the phases to establish the BOD:COD and BOD:NH₃-N ratios would have saved considerable lab time.

Heterotrophic plate counting (HPC), while useful for establishing the presence of active bacterial populations, does not lend itself well to comparison with the other tests. Either such a test could be omitted, or could be conducted using selective media. For example, a specific organic carbon consumer and a specific nitrifier could be tracked via HPC to monitor when nitrifiers begin to assume dominance over carbon consumers in the reactors.

Clogging of landfill leachate collection systems has been a subject of much research in recent years. The recirculation of leachate into the landfill after passing through a simple treatment step such as a trickling filter could help to remove some of the ionic and biological content in the leachate, thereby diluting its strength as it passes through the landfill. Therefore, future research work encompassing a pilot-scale experiment to test this approach would be useful. In addition, further research is required into the use of MBG in a variety of landfill applications, which could represent significant additional opportunities to reuse this waste material.

The broad objective of this project is to find a low-cost method to treat landfill leachate, in order to help encourage the development of needed landfill capacity. From the point of view of this objective, several policy recommendations can be suggested. Firstly, the province of Ontario should develop an official waste management plan, as other provinces have done already. One plank in a waste management plan could be a requirement for communities (with appropriate exceptions) to be "waste-independent"; that is, to be able to provide for their own waste disposal. With sufficient provincial support this would enable communities to either develop their own facilities or enter into cooperative agreements with like communities, rather than simply paying someone else to take out the trash. This sort of policy would address the capacity crisis at its root. Secondly, a new landfill inventory should be created; the last landfill inventory was conducted in 1991 and is sorely out of date. An updated inventory would quantify the province's available and potential capacity, enabling policymakers to prioritize how to invest scarce funding in this necessary part of the province's infrastructure.

References

- Amari T, Themelis NJ, Wernick IK. 1999. Resource recovery from used rubber tires. *Resources Policy* 25(3):179-88.
- Anon. 1994. Waste glass serves as road aggregate. *Public Works* 125(6):52.
- [Arkansas] Arkansas Dept. of Environmental Quality Solid Waste Management Division. 2003. IPPC for the use of shredded tires in construction of landfill drainage media. Little Rock, Ar: Arkansas Dept. of Environmental Quality [cited 5/5/2009]. Available from: http://www.adeq.state.ar.us/solwaste/branch_technical/pdfs/30-Shredded_Tire_IPPC_031125.pdf.
- ASTM. ASTM D6270-08 standard practice for use of scrap tires in civil engineering applications [Internet]; c2009 [cited 3/13/2009]. Available from: <http://www.astm.org/Standards/D6270.htm>.
- Ball HL. 1998. Optimizing the performance of sand filters and packed bed filters through media selection and dosing methods. Biosolutions Incorporated [cited 5/5/2009]. Available from: <http://www.biosolutions.org/pdf/Optimizing-Filter-Performance.pdf>.
- Benda E. 2008 May 13. Re: Leachate composition report. Personal email.
- Bilgili MS, Demir A, Akkaya E, Ozkaya B. 2008. COD fractions of leachate from aerobic and anaerobic pilot scale landfill reactors. *Journal of Hazardous Materials* 158(1):157-63.
- Blackman A and Palma A. 2002. Scrap tires in ciudad juarez and el paso: Ranking the risks. *Journal of Environment and Development* 11(3):247-66.
- The Bore-lee Glass Equipment. Beveller, Glazing Machinery, Glass Blowing Equipment, Used Glass Industry Equipment - Bor-lee Glass Equipment [Internet]. USA: Borlee Used Glass Machinery; c2009 [cited 5/5/2009]. Available from: <http://www.borlee.net/cgi-bin/rexview.cgi?rex=000745&wsc=22-1709>.
- Clabaugh MM. 2001. Nitrification of landfill leachate by biofilm columns. Blacksburg, Virginia: Virginia Polytechnic Institute and State University.
- Clesceri LS, Greenberg AE, Eaton AD, editors. 1998. Standard methods for the examination of water and wastewater. Washington D.C.: American public health association. 20th ed. New York: American Public Health Association; Water Environment Federation.
- Cummings B. 2008 Dec 16. By telephone.

- [CWC] Clean Washington Center. 1997. Crushed glass as a filter medium for the on-site treatment of wastewater. Seattle, Wa: Pacific North West Economic Region. Technology Brief #GL-97-2 [cited 5/5/2009]. Available from: www.cwc.org/glass/gl972fs.pdf.
- Dahme H and Lococo T. 2003. Old ghosts: Closed municipal landfill sites and abandoned industrial sites. Toronto: Gowling Lafleur Henderson LLP [cited 5/5/2009]. Available from: www.aboutremediation.com/PDFS/Closed_Mun_Landfills_and_Ind_Sites_OBA_2003_Gowlings.pdf.
- Davies G. and Kamp A. 2005. Site closure process and alternatives for waste disposal. Landfill sites in northwestern ontario: Issues and opportunities workshop; 2005 Mar 2; Thunder Bay.
- Del Pozo R and Diez V. 2003. Organic matter removal in combined anaerobic-aerobic fixed-film bioreactors. *Water Research* 37(15):3561-8.
- Dermou E and Vayenas DV. 2007. A kinetic study of biological Cr(VI) reduction in trickling filters with different filter media types. *Journal of Hazardous Materials* 145(1-2):256-62.
- Diamadopoulos E. 1994. Characterization and treatment of recirculation-stabilized leachate. *Water Research* 28(12):2439-45.
- Eash N, Green CJ, Ravzi A, Bennett WF, Bratz MC. 2008. *Soil science simplified*. 5th ed. New York, NY: Wiley-Blackwell.
- Edgehill RU. 1996. Degradation of pentachlorophenol (pcp) by arthrobacter strain atcc 33790 in biofilm culture. *Water Research* 30(2):357-63.
- Edil TB, Park JK, Kim JY. 2004. Effectiveness of scrap tire chips as sorptive drainage material. *Journal of Environmental Engineering* 130(7):824-31.
- [Edmonton] Edmonton Waste Management Centre of Excellence. 2007. Landfill methane oxidation – project overview. Edmonton, AB: Edmonton Waste Management Centre of Excellence [cited 5/5/2009]. Available from: http://ewmce.com/images/stories/PDF_Files/04-07R.pdf.
- Elliott RW. 2001 Jul. Filtration: Evaluation of the use of crushed recycled glass as a filter medium part 1. Accessed 2008 Dec 1. *Water Engineering & Management*: 13-16.
- Farquhar GJ. 1989. Leachate: Production and characterization. *Canadian Journal of Civil Engineering* 16(3):317-25.

Gálvez JM, Gómez MA, Hontoria E, González-López J. 2003. Influence of hydraulic loading and air flowrate on urban wastewater nitrogen removal with a submerged fixed-film reactor. *Journal of Hazardous Materials* 101(2):219-29.

Grimes BH, Steinberk S, Amoozegar A. 2003. "Analysis of tire chips as a substitute for stone aggregate in nitrification trenches of on-site septic systems: Status and notes on the comparative microbiology of tire chip versus stone aggregate trenches". *Small Flows Quarterly* 4(4).

[HRM] Halifax Regional Municipality. HRM - Waste Water Treatment - Leachate Treatment Facility [Internet]. Halifax, NS: Halifax Regional Municipality; c2005 [cited 2009 May 7]. Available from: <http://www.halifax.ca/Works/wwt/leachate.html>.

Haydar MM and Khire MV. 2007. Leachate recirculation using permeable blankets in engineered landfills. *Journal of Geotechnical and Geoenvironmental Engineering* 133(4):360-71.

Horan NJ and Lowe M. 2007. Full-scale trials of recycled glass as tertiary filter medium for wastewater treatment. *Water Research* 41(1):253-9.

Hrapovic L. 2001. Biodegradation of organic contaminants under diffusive transport through clayey soil. Ph.D thesis. London, On: The University of Western Ontario.

[IPPC] Integrated Pollution Prevention and Control (UK). 2007. Guidance for the treatment of landfill leachate. UK: Integrated Pollution Prevention and Control (UK). Sector Guidance Note IPPC S5.03 [cited 5/5/2009]. Available from: <http://www.environment-agency.gov.uk/research/library/consultations/36966.aspx>.

Jokela JPY, Kettunen RH, Sormunen KM, Rintala JA. 2002. Biological nitrogen removal from municipal landfill leachate: Low-cost nitrification in biofilters and laboratory scale in-situ denitrification. *Water Research* 36(16):4079-87.

Kim JY, Park JK, Edil TB. 1997. Sorption of organic compounds in the aqueous phase onto tire rubber. *Journal of Environmental Engineering* 123(9):827-34.

Knox K. 1985. Leachate treatment with nitrification of ammonia. *Water Research* 19(7):895-904.

[LaFlèche] LaFlèche Environmental Inc. 1999. LaFlèche Eastern Ontario waste handling facility. Moose Creek, ON: LaFlèche Environmental Inc. Rev 0. Section 2848-6 p. Available from: Specification provided by Bray, T. via email, 2008 May 22.

LeDuc M. 2008. In-person conversation.

- Mann A, Mendoza-Espinosa L, Stephenson T. 1998. A comparison of floating and sunken media biological aerated filters for nitrification. *Journal of Chemical Technology and Biotechnology* 72(3):273-9.
- Martin EJ, Martin ET. 1991. *Technologies for small water and wastewater systems*. New York: Van Nostrand Reinhold.
- McBean EA, Rovers FA, Farquhar GJ. 1995. *Solid waste landfill engineering and design*. Englewood Cliffs, N.J.: Prentice Hall PTR.
- McKenzie CM. 2003. Tire chips: A growing trend as aggregate in soil absorption systems. *Small Flows Quarterly* 4(4):14-7.
- Merritt NJ, Redmond WH, Pearson MM. 2005. Government and market mechanisms to provide alternatives to scrap tire disposal. *Journal of Nonprofit & Public Sector Marketing* 13(1,2):151.
- Metcalf & Eddy. 2002. *Wastewater engineering: Treatment and reuse*. 4th ed. revised by Tchobanoglous G, Burton FL, Stensel HD. Toronto: McGraw-Hill.
- Miller G. 2008. Getting to k(no)w: The 2007 – 2008 annual report of the Environmental Commissioner of Ontario. Toronto: Environmental Commissioner of Ontario [cited 2009 5/5/2009]. Available from: <http://www.eco.on.ca/eng/index.php/pubs/eco-publications/2007-08-AR.php>.
- Miller G. 2006. Neglecting our obligations: The 2005 – 2006 annual report of the Environmental Commissioner of Ontario. Toronto: Environmental Commissioner of Ontario [cited 5/5/2009]. Available from: <http://www.eco.on.ca/eng/index.php/pubs/eco-publications/2005-06-annual-report.php>.
- [Mississippi] Mississippi Department of Environmental Quality Solid Waste Management Branch. 2002. Beneficial use of waste tire material: IPPC for using tire chips as leachate drainage material at municipal solid waste landfills. Jackson, MS: Mississippi Department of Environmental Quality [cited 5/5/2009]. Available from: [http://www.deq.state.ms.us/mdeq.nsf/pdf/SW_WasteTireUses/\\$File/WasteTireUses.pdf?OpenElement](http://www.deq.state.ms.us/mdeq.nsf/pdf/SW_WasteTireUses/$File/WasteTireUses.pdf?OpenElement).
- [Missouri] Missouri Department of Natural Resources. Cleanup begins at Missouri's largest remaining tire dump. News release no. 196 [Internet]. Jefferson City, Mo: Missouri Department of Natural Resources; c2006 [cited 2008 Dec 1]. Available from: http://www.dnr.mo.gov/newsrel/nr06_196.htm.
- [MOE] Ontario Ministry of the Environment. 1991. Waste disposal site inventory. Toronto: Queen's Printer for Ontario. Report nr PIBS 256. 1 p. Available from: Ministry of the Environment Public Information Centre 1-800-565-4923.

- [MOE] Ontario Ministry of the Environment. Ministry of the Environment - Land - Landfills [Internet]; c2008 [cited 3/12/2009]. Available from: <http://www.ene.gov.on.ca/en/land/wastedisposal/index.php#incinerators> .
- [MOE] Ontario Ministry of the Environment. Environmental Registry [Internet]. Toronto, On: Queen's Printer for Ontario; c2009 [cited 4/5/2009]. Available from: <http://www.ebr.gov.on.ca/ERS-WEB-External/displaynoticecontent.do?noticeId=MTA1OTc3&statusId=MTU4ODk1&language=en>.
- Mondal B and Warith MA. 2008. Use of shredded tire chips and tire crumbs as packing media in trickling filter systems for landfill leachate treatment. *Environmental Technology* 29(8):827-36.
- Mondal B. 2006. Use of shredded tire chips and tire crumbs as packing media in trickling filter system for landfill leachate treatment. M.A.Sc thesis. Ryerson University: Toronto, On.
- [NSRA] Nebraska State Recycling Association. 2001. Recycling tires: Tire chip utilization for landfill leachate collection layer. Accessed 2008 Dec 1. Lincoln, Ne: Nebraska State Recycling Association [cited 5/5/2009]. Available from: <http://www.p2pays.org/ref/15/14315.pdf>.
- [OECD] Organisation for Economic Co-operation and Development. 2006. Improving recycling markets. Paris and Washington, D.C.: Organisation for Economic Co-operation and Development.
- Onay TT and Pohland FG. 1998. In situ nitrogen management in controlled bioreactor landfills. *Water Research* 32(5):1383-92.
- [OTS] Ontario Tire Stewardship Board of Directors. 2004. Scrap tire diversion program for Ontario obtains approval. Toronto: Ontario Tire Stewardship [cited 5/5/2009]. Available from: www.ontariotirestewardship.org/PDFFiles/OTSPressReleaseSept272004.pdf.
- [OTS] Ontario Tire Stewardship [Internet]. Toronto: Ontario Tire Stewardship; c2009 [cited 5/01/2009]. Available from: <http://www.ontariotirestewardship.org/> .
- [OWMA] Ontario Waste Management Association. 2006. A landfill story: Capacity, export & EA. Available from: <http://www.owma.org/db/db2file.asp?fileid=457>, Accessed 2009 Mar 12.
- [PacWest] Pacific West Corporation. Plastic products by Pacific West Corporation [Internet]: Pacific West Corporation; c2009 [cited 3/12/2009]. Available from: <http://www.pacwestcorp.com.au/>.

- Park JM, Choi CY, Seong BL, Han MH. 1982. Effect of mass transfer in a recirculation batch reactor system for immobilized penicillin amidase. *Biotechnology and Bioengineering* 24(10):2215-26.
- Phaneuf R. and Glander C. 2003. Using tire chips in landfill leachate collection and removal systems under 6 NYCRR part 360. Federation of New York Solid Waste Associations 2003 solid waste/recycling conference & trade show; 2003 May 7; Bolton Landing, NY [cited 5/5/2009]. Available from: http://www.nyfederation.org/PDF2003/26_federation%20tire%20chips%20in%20lcrss%202003.pdf; New York State Department of Environmental Conservation Division of Solid & Hazardous Materials.
- Reynolds TD and Richards PA. 1996. Unit operations and processes in environmental engineering. 2nd ed. Boston: PWS Pub. Co.
- Robinson HD and Maris PJ. 1983. The treatment of leachates from domestic wastes in landfills. I. aerobic biological treatment of a medium-strength leachate. *Water Research* 17(11):1537-48.
- Rowe R. K. 1995. Leachate characteristics for MSW landfills. in: *Proceedings of Sardinia 95 – 5th international landfill symposium*; 1995; Cagliari, Italy: . pp. 327–344 p.
- Rowe RK and McIsaac R. 2005. Clogging of tire shreds and gravel permeated with landfill leachate. *Journal of Geotechnical and Geoenvironmental Engineering* 131(6):682-93.
- Rowe RK, VanGulck JF, Millward SC. 2002. Biologically induced clogging of a granular medium permeated with synthetic leachate. *J. Environ. Eng. Sci.* 1:135-56.
- Rowe RK, Armstrong MD, Cullimore DR. 2000. Mass loading and the rate of clogging due to municipal solid waste leachate. *Canadian Geotechnical Journal* 37(2):355-70.
- [RRI] Recycling Research Institute. Scrap Tire News Online: the scrap tire and rubber industry portal | article [Internet]: Recycling Research Institute; c2002 [cited 3/13/2009]. Available from: <http://www.scraptirenews.com/99apr3.html>.
- Ruffier PJ, Boyle WC, Kleinschmidt J. 1981. Short-term acute bioassays to evaluate ammonia toxicity and effluent standards. *Journal of the Water Pollution Control Federation* 53(3 I):367-77.
- Schmucker Bruce O. and Buffalini Rick J. 1995. Using pulverized glass as a landfill liner system protective cover and leachate drainage layer. 388 p.

- Shin H-S, Yoo K-S, Park JK. 1999. Removal of polychlorinated phenols in sequential anaerobic-aerobic biofilm reactors packed with tire chips. *Water Environment Research* 71(3):363-7.
- Shipin OV, Lee SH, Chiemchaisri C, Wiwattanakom W, Ghosh GC, Anceno AJ, Stevens WF. 2007. Piggery wastewater treatment in a tropical climate: Biological and chemical treatment options. *Environmental Technology* 28(3):329-37.
- Soares A, Guieysse B, Mattiasson B. 2003. Biodegradation of nonylphenol in a continuous packed-bed bioreactor. *Biotechnology Letters* 25(12):927-33.
- Spears J. 2008 Mar 4. Glass recycling plan to save city millions. *The Toronto Star*; Available from: <http://www.thestar.com/News/GTA/article/309018>; Accessed 2009 May 5.
- Statistics Canada. 2005. Human activity and the environment annual statistics 2005 feature article: Solid waste in Canada. Ottawa: Minister of Industry. Report nr 16-201-XIE. 1 p. [cited 5/5/2009]. Available from: <http://dsp-psd.tpsgc.gc.ca/Collection-R/Statcan/16-201-XIE/16-201-XIE.html>.
- Stewardship Ontario. 2007. Stewardship Ontario annual report 2007. Toronto: Stewardship Ontario [cited 5/5/2009]. Available from: http://www.stewardshipontario.ca/corporate/annual_report.html.
- Tashiro T. 2007. Tafaigata landfill rehabilitation project. Apia, Samoa: Secretariat of the Pacific Regional Environment Programme [cited 5/5/2009]. Available from: 221.114.113.155/samoa/SWMW/swm-workshop/2007-06-Fiji/fiji-2007-6-14/frank/Tafaigata_Treatment_design.pdf.
- [Thunder Bay] Corporation of the City of Thunder Bay. Proposed John Street Landfill expansion [Internet]. Thunder Bay, ON; c2008 [cited 2008 Dec 24]. Available from: <http://www.thunderbay.ca/index.cfm?fuse=html&pg=3860>.
- [Toronto] City of Toronto. City of Toronto: Solid Waste Management - Waste reduction tips [Internet]; c2009 [cited 3/12/2009]. Available from: http://www.toronto.ca/garbage/green_lane.htm#key.
- Villeneuve P. 2008 Dec 16. By telephone.
- Warith M. and Rao S. M. 2004. Estimation of maximum leachate head in tire shred drainage layer. 167 p.
- Warith MA and Rao SM. 2006. Predicting the compressibility behaviour of tire shred samples for landfill applications. *Waste Management* 26(3):268-76.

- Warith MA, Evgin E, Benson PAS, Rao SM. 2005. Evaluation of permeability of tire shreds under vertical loading. *Journal of Testing and Evaluation* 33(1):51-4.
- Waste Management. Glanbrook Landfill [Internet]; c2009 [cited 3/12/2009]. Available from: <http://www.wm.com/Templates/FAC4922/contacts.asp>.
- Wellington A. 2007. ES 8921 environmental law and policy course manual - Ryerson University. Toronto: Captus Press.
- Xie W, Wang Q, Song G, Kondo M, Teraoka M, Ohsumi Y, Ogawa HI. 2004. Upflow biological filtration with floating filter media. *Process Biochemistry* 39(6):765-70.
- Yang Z. 2002. Biological wastewater treatment: Pilot-plant studies using a trickling filter. Toronto: Ryerson University.
- Zecca J. 2006. Madison County glass recycling program. Federation of NYS waste associations solid waste/recycling conference & trade show; 2006 May 8; Madison County, NY [cited 5/5/2009]. Available from: http://www.nyfederation.org/pdf2006/58_m8Zecca.pdf: Madison County Department of Solid Waste.